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Ecological Risk Assessment for the Middle Snake River, Idaho

National Center for Environmental Assessment—Washington Office
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC

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U.S. Environmental Protection Agency

National Center for Environmental Assessment-Washington Office
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Office of Environmental Assessment
Region 10
Seattle, Washington

DISCLAIMER

ABSTRACT

An ecological risk assessment was completed for the Middle Snake River, Idaho. In this assessment, mathematical simulations and field observations were used to analyze exposure and ecological effects and to estimate risk.

The Middle Snake River which refers to a 100 km stretch (Milner Dam to King Hill) of the 1,667 km long Snake River lies in the Snake River Plain of southern Idaho. The contributing watershed includes 22,326 square km of land below the Milner Dam and adjacent to the study reach. The demands on the water resources have transformed this once free-flowing river segment to one with multiple impoundments, flow diversions, significant alterations to river habitat, loss of native macroinvertebrate species, extirpation of native fish species, expansion of pollution-tolerant organisms, and excessive growth of macrophytes and algae.

The environmental management goals for this assessment are: *"attainment of water quality standards, establishment of total maximum daily loads for major pollutants, water for hydropower, recreation, and irrigation, recovery of endangered species, and sustained economic well being*." The diversity, reproduction, growth, and survival of representative species from three major trophic levels (fish, invertebrates, and plants) were chosen as assessment endpoints in order to complete an ecosystem level analysis.

Simulation of habitat conditions (temperature, water velocity, and water depth) and review of field studies show that most spawning, rearing, and adult habitats available to native fish species and in the Middle Snake River are undesirable. In addition to high water temperatures, our analysis showed that low flows and sedimentation are main stressors affecting these fish species. These same factors are thought to be responsible for the decline of native snail populations. Risks of eutrophication were estimated by changes in the plant biomass. The simulation of macrophyte growth, under existing conditions in the study reach, indicates the river is eutrophic based on aquatic plant biomass exceeding 200 g/m^2. The lines of evidence drawn from the model simulation suggest that nutrients, temperature, flow, and water depth are the major factors controlling macrophyte growth.

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FOREWORD

Risk assessment is playing an increasingly important role in determining environmental policies and decisions at the U.S. Environmental Protection Agency (EPA). EPA's first Agency-wide guidelines for ecological risk assessment, published in May 1998, provided a broad framework applicable to a range of environmental problems associated with chemical, physical, and biological stressors. As ecological risk assessment evolves, it is moving beyond a focus on single species toward addressing multiple species and their interactions, and from assessing effects of simple chemical toxicity to the cumulative impacts of multiple interacting chemical, physical, and biological stressors on species, populations, communities, and ecosystems in watersheds, regions, or other "places."

To further develop and demonstrate the use of the ecological risk assessment paradigm in addressing such environmental problems, the EPA sponsored and is completing four watershed assessments, including this one on the Middle Snake River. Ecological risk assessments, when applied to watersheds, must be adaptable to a lack of complete knowledge about complex ecosystem dynamics and the need to reach consensus in watershed groups. Thus, watershed ecological risk assessments may not characterize every structural and functional element of the ecosystem and may be limited by management constraints. The strengths of applying ecological risk assessment and adaptations that need to be made to implement the approach in watersheds are discussed in other EPA reports and in scientific journal articles and conference proceedings (see www.epa.gov/ncea). The other three assessments are also presented elsewhere.

The Middle Snake River was selected as one of these watershed assessments because of its unique species which are threatened or endangered, the multiple stressors, and the related concerns of interested citizens, government institutions, and private industry. This assessment is intended to address such concerns by analyzing the Middle Snake River's stressors and resulting ecological effects and to stimulate broader public awareness and participation in decision making for reducing ecological risks. This watershed assessment report serves as an example of how ecological risk assessment principles can be applied at the watershed scale to improve the use of science in decision making.

Michael Slimak
Associate Director of Ecology
EPA, National Center for Environmental Assessment

PREFACE

The National Center for Environmental Assessment–Washington Office (NCEA–W) provided document production, printing, and distribution support for this document. Funding was provided by EPA's Office of Water, Region 10, and the State of Idaho Department of Environmental Quality (IDEQ).

This document presents the ecological assessment of several years of monitoring data as well as a simulation of present and future conditions of the river. The idea for this assessment began with EPA Region 10 managers working closely with managers from IDEQ. They had a vision for a comprehensive assessment of the Middle Snake River. Their goal was a restoration of this system for the people who live in the area as well as the aquatic organisms that inhabit the river.

The initial work for this study began in 1987, with river monitoring work done by the IDEQ. The EPA was invited to participate in the design of this data collection and ultimately the risk assessment by the Middle Snake River County Planning Group. Thus, genesis of the ecological risk assessment began before EPA had completed its *Guidelines for Ecological Risk Assessment* in 1998. Additionally, as discussed in the Foreword, ecological risk assessments implemented in watersheds need to be flexible due to data limitations and client needs. Although there may be a few inconsistencies, the process for completing the final assessment and this report were based on the EPA guidelines and advice and support from NCEA–W.

This risk assessment provided decision makers and interested citizens with factual information for their deliberations. They are able to draw on components of the analysis as well as the conclusions of the assessment for their actions on the river. The results of the assessment were used by the IDEQ in developing their Nutrient Management Plan and Total Maximum Daily Loads (TMDL) for pollutant discharge permits on the river.

This final document reflects a consideration of all comments received during internal and external peer review provided by a number of experts between 1996-2001.

AUTHORS, CONTRIBUTORS, AND REVIEWERS

Authors

Patricia A. Cirone
Duane W. Karna
John R. Yearsley
U.S. Environmental Protection Agency
Office of Environmental Assessment
Region 10
Seattle, WA

C. Michael Falter
University of Idaho
Moxcow, ID

Todd V. Royer
University of Illinois
Urbana, IL

Contributors

Dr. Gerald Filbin
U.S. Environmental Protection Agency
Washington, DC

Dr. Suzanne Marcy
U.S. Environmental Protection Agency
NCEA
Anchorage, AK

Victor Serveiss
U.S. Environmental Protection Agency
NCEA–W
Washington, DC

Dr. Michael Watson
U.S. Environmental Protection Agency
Region 10
Seattle, WA

Reviewers

Jim Andreason
U.S. Environmental Protection Agency
NCEA–W
Washington, DC

Peter Bowler
University of California, Irvine
Irvine, CA

Kellie Kubena
U.S. Environmental Protection Agency
Region 10
Seattle, WA

Wayne Minshall
Idaho State University
Pocatello, ID

Geoffrey Poole
U.S. Environmental Protection Agency
Region 10
Seattle, WA

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- Vic Serveiss ensured compliance with the Guidelines for Ecological Risk Assessment and consistency with other watershed ecological risk assessments.

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- The Middle Snake River County Planning group for the impetus to begin this assessment and their valuable insights during the initial discussions about the problems in the Middle Snake River.

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- Dr. Peter Bowler and Dr. Wayne Minshall provided advice in designing this study, valuable information throughout the process, and excellent review comments.

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We would like to dedicate this report to Dr. Tim Litke, IDEQ, who was the inspiration for many of us. He brought a diverse group of people together and established a common goal. We hope that the restoration of this ecosystem will be the memorial to his special contribution.

1. EXECUTIVE SUMMARY

1.1. INTRODUCTION

This report presents the results of an ecological risk assessment for the Middle Snake River, Idaho. Ecological risk assessment is a process for analyzing and presenting information on the risks of exposures of organisms, populations, communities, or ecosystems to stressors and disturbances. The assessment process (Figure 1-1) begins with planning and problem formulation, proceeds through analysis of exposure and effects, and ends with risk characterization. During the planning phase the management goals are established for the watershed. In problem formulation, the watershed is described, ecologically relevant assessment endpoints are defined, and a conceptual model is developed. In this assessment, mathematical simulations and observations are used to analyze exposure and ecological effects and to estimate risk. Finally, in risk characterization, all the elements of the assessment are brought together to reach a conclusion. These elements include an uncertainty analysis, lines of evidence supporting the risk estimate(s), and the likelihood of recovery.

The Snake River is the tenth longest river in the United States, extending 1,667 km from its origins in western Wyoming to its union with the Columbia River at Pasco, Washington.

The river reach of concern (Milner Dam to King Hill), hereafter referred to as the Middle Snake River, spans roughly 150 km and lies in the Snake River Plain of southern Idaho. The contributing watershed includes 22,326 square km of land below the Milner Dam and adjacent to the study reach.

The demands on the water resources have transformed this once free-flowing river segment to one with multiple impoundments, flow diversions, and increased chemical and microbiological pollutant loadings. The Snake River has long been valued as a resource of water for irrigation for hydropower. Physical changes include significant alterations to rapids and pool areas of the river. Prior to impoundment of the river, chinook salmon were able to migrate as far as Shoshone Falls, a natural barrier. Resulting biological changes include loss of native macroinvertebrate species, invasion and dominance by exotic species, extirpation of native fish species, expansion of pollution-tolerant organisms, and excessive growth of aquatic plants and algae. The increasing demand for energy, irrigation resources, springs, and dairy feedlots projected for this region will place additional burdens on an ecosystem that human activity has already substantially changed.

The U.S. Environmental Protection Agency sponsored five watershed ecological risk assessment case studies projects across the country. The purpose is to learn how to develop ways to analyze, characterize, and communicate the severity of ecological risk to valued

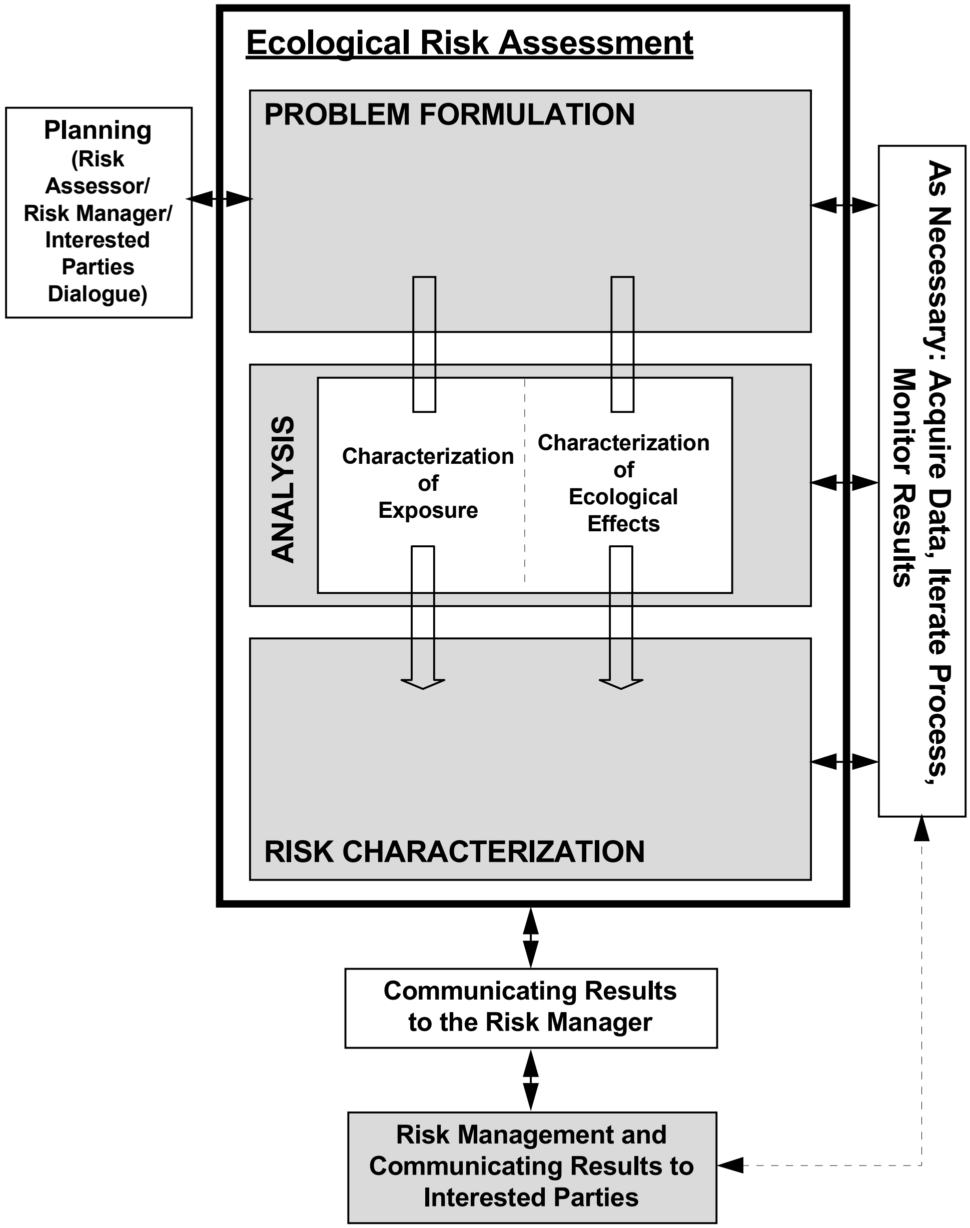

Figure 1-1. Framework for ecological risk assessment.

environmental resources. The Middle Snake River was selected as one of the five watershed case studies because of its unique species that are threatened or endangered, the multiple stressors, and the concerns of interested citizens, government institutions, and industry individuals. This ecological risk assessment was undertaken to address such concerns by analyzing the Middle Snake River's stressors and resulting ecological effects, and to stimulate broader public awareness and participation in decision making for reducing ecological risks.

1.2. PLANNING

During the planning phase of this assessment a number of Federal, State, and county organizations, along with private organizations, academic researchers, and interested citizens, participated in workshops and meetings to discuss long-term goals for the river. The management goals that were identified in this process were:

> *attainment of water quality standards, establishment of total maximum daily loads for major pollutants, water for hydropower, recreation, and irrigation; recovery of endangered species, and sustained economic well being*.

1.3. PROBLEM FORMULATION

Representative species from three major trophic levels (fish, invertebrates, and plants) were chosen as assessment endpoints to complete an ecosystem-level analysis that would provide information for the public and decision makers. Each group is an important link in the structure and function of this riverine ecosystem. It was determined that analysis of the factors controlling the species' functions (reproduction, growth, and survival) should provide evidence for the primary causes of the ecosystem changes.

This risk analysis began with the identification of the ecosystem driving forces (hydrology, hydraulics, meteorology, and land-use activities) that define the structure and function of this ecosystem. The land-use activities were superimposed on the natural system. Irrigated agriculture, aquaculture, cattle feeding lots, sewage treatment, and impoundments were found to be the major land use activities in the watershed. The hypothesis for this analysis is that materials (nutrients, sediments, thermal energy, and ammonia) released from these activities, as well as the habitat alterations resulting from impoundments, can produce stressful conditions that are harmful to the native aquatic biota. Stressful conditions result from altered water velocity and depth, decreased dissolved oxygen, increased water temperature, disrupted sedimentation, excessive nutrient loading, and increased eutrophication.

1.4. ANALYSIS

A mathematical model was developed for the river reach of concern for the period January 1990 to December 1994. Ecosystem driving forces and land-use activities were combined in the model to describe the ecosystem dynamics. The dynamics of the ecosystem were simulated with variables including meteorological conditions, hydrological and hydraulic conditions, carbonaceous biological oxygen demand, dissolved oxygen, phytoplankton biomass, organic nitrogen, ammonia nitrogen, nitrite and nitrate nitrogen, organic phosphorus, orthophosphorus, temperature, coliform bacteria, water depth, water velocity, rooted aquatic plants, epiphytes, and periphyton. Stressor characteristics are defined in terms of probability models for point-source loadings and nonpoint-source loadings.

Risks for fish and macroinvertebrates were estimated by determining the likelihood of being above or below cold-water biota tolerance limits. Tolerance limits are generally the natural levels to which most native species have adapted. Excursions above and below these boundaries or tolerance levels can be stressful. The tolerance limits for fish and macroinvertebrates were based on the State of Idaho Water Quality Standards for temperature, dissolved oxygen, total phosphorus, and ammonia. In addition to comparison with water quality standards, the risks to fish species were estimated by determining the likelihood of the river supporting suitable habitats for their reproduction and survival. These habitat limits are based on Habitat Suitability Indices developed by the U.S. Fish and Wildlife Service. The indices used in this risk assessment were based on temperature, water depth, and water velocity preferences for fish species of interest.

Risks of eutrophication were estimated by changes in plant biomass. The State of Idaho defines “nuisance” as unacceptable for plant biomass. For the purposes of this analysis, “nuisance” was defined as exceeding a biomass level (200 g/m^2) found in eutrophic systems.

Finally, a qualitative analysis of data from field surveys in the Middle Snake River, literature reviews of other studies, and best professional judgment were discussed as additional lines of evidence for factors controlling the fish, macroinvertebrate, and aquatic plant populations.

The quantitative measures of effect included water quality standards and habitat suitability indices. The water quality standards that were used to define the level of concern for temperature, dissolved oxygen, ammonia, and macrophytes were limited in the breadth of applicability. The standards were expressed as discrete numbers rather than distributions. As discrete numbers they did not necessarily reflect the specific requirements of individual species. The habitat suitability indices are based primarily on water depth and water velocity; thus the effects of biological interactions or substrate morphology are not taken into account. These other factors could have significant effects on fish populations.

1.5. RISK CHARACTERIZATION

The results of mathematical simulations of a 67-year record for daily average and maximum levels of dissolved oxygen and temperature at 13 distinct river locations were compared to the water quality criteria for cold-water biota, spawning whitefish, and spawning rainbow trout. The cold-water biota limits were seldom exceeded (less than 1% of the time). However, the frequency of not attaining dissolved oxygen limits in certain locations for spawning mountain whitefish or rainbow trout ranged from 1% to 57% depending on the location and timing of spawning. Further, water temperatures needed for spawning and incubation are essentially absent for these fish. Ammonia did not appear to be a major stressor in this ecosystem. The likelihood of exceeding chronic or acute ammonia tolerance limits was less than 5% and 1%, respectively.

Our simulation of habitat conditions (temperature, water velocity, and water depth) and review of field studies also show that most spawning, rearing, and adult habitats available to rainbow trout, mountain whitefish, and white sturgeon in the Middle Snake River are undesirable. In addition to high water temperatures, our analysis showed that low flows and sedimentation are main stressors affecting these fish species. These adverse conditions can be improved if a spring freshet is reestablished with flows large enough for successful spawning, and with post-spawning water temperatures low enough to allow for healthy embryonic development other than during a narrow window of time.

Analysis of ecological effects for macroinvertebrates was made primarily by inference. Water quality standards for cold-water biota were assumed to represent conditions favorable to invertebrate growth and survival. In most cases, the likelihood of falling below these limits was low throughout the river. This result implies that the temperature and dissolved oxygen levels did not exceed tolerance limits for macroinvertebrates. The evidence from field surveys conflicts with the standards set for cold-water biota. The decline of native snails in this reach suggests that the temperature, dissolved oxygen, and physical habitat changes are in fact detrimental to the survival, reproduction, and diversity of the snails. Their life history information indicates that they prefer colder temperatures, more swiftly flowing water, and higher dissolved oxygen than are allowed for in the standards.

The simulation of macrophyte growth under existing conditions in the study reach indicates the river is eutrophic because aquatic plant biomass exceeds 200 g/m^2. The lines of evidence drawn from the model simulation suggest that nutrients, temperature, flow, and water depth are the major factors controlling macrophyte growth. The evidence for phosphorus as a limiting factor is derived from model simulations. There is a 23% to 25% likelihood that phosphorus will be equal to or less than the State of Idaho's limit of 0.075 mg/L in the upper reaches of the river. The inflow of large volumes of spring water with low levels of phosphorus decreases the likelihood to 7% to 18%. There is only a 1% chance the macrophyte biomass will be less than 200 g/m^2.

However, this evidence does not in and of itself define all the factors controlling the excess growth of aquatic plants. In fact, it is the field surveys that provide the additional evidence to describe control of plant growth. Water chemistry data show that nitrogen as well as phosphorus can be limiting, depending on the volume and quality of inflow to the river. Thus, the nutrient limits will shift from nitrogen to phosphorus during seasons and years. In addition, the physical characteristics of the water column and bottom substrate are critical factors in limiting growth. On the basis of field observations, the combination of deep, fine, nutrient-rich sediments downstream of those areas in the Middle Snake River receiving organic and nutrient loading favors aquatic plant growth. Once beds develop in these regions, internal water velocities slow, resulting in further sediment deposition. These reduced velocities and relatively clear waters provide optimum conditions for increased plant growth. These sediments provide an anchor for rooted vascular plants, which in turn provide the habitat for nonrooted plants. In addition to a substrate for growth, the sediments store nutrients vital to the growth of rooted aquatic plants. This cycle is borne out in numerous areas throughout the river. The evidence is strengthened by the high flow season in 1997. With the rush of water, the sediments were flushed and the macrophytes did not develop to levels seen in previous years when flows were much lower.

Uncertainty in this assessment includes variability in ecosystem driving forces and stressors, sources of mass and energy, model error, parameter estimation error, measurement error, errors in measures of effect, and lack of knowledge. Variability in estimates of stressors (loading from land-use activities) was expressed as cumulative distributions with error bars. Variability in the ecosystem driving forces (hydrology and meteorology) was based on the actual and adjusted 67-year record, respectively.

Bias in the model was examined by comparing the simulated results with field data collected from the Middle Snake River. The best correlation was found between temperature and nitrate-nitrite nitrogen. The correlation of simulated and observed dissolved oxygen levels was only partially good, implying that the model may not accurately predict primary productivity. Correlation of simulated and observed total phosphorus and total ammonia nitrogen was low. The model predicted higher values of ammonia and underpredicted total phosphorus. The high levels of ammonia may be due to poor loading estimates for the sewage treatment plant or other biological interactions. The underprediction of phosphorus also may be due to poor loading estimates or loss of phosphorus through sediment and plant sequestration.

Uncertainty due to lack of knowledge will result in errors of judgment as well as model errors. In particular, the lack of species-specific information for the snails endemic to the Middle

Snake River makes it difficult to confirm a definitive cause for ecological effects on these organisms. However, the evidence for rainbow trout, mountain whitefish, sturgeon, and other cold-water biota can be used by inference to bolster the argument for snails.

The processes of eutrophication and habitat alteration in the Middle Snake River are driven by a series of changes. It is clear that these cannot be attributed to any one factor. It is therefore difficult to define management options that would foster an easy recovery. The influence of increased water flows on system recovery was clearly demonstrated in 1997; the movement of sediments, nutrients, and macrophytes was dramatic. However, the macroinvertebrate and fish populations are not going to recover after 1 year of high flows. Flow and sedimentation processes must return to a more natural regime before the aquatic populations will rebound.

2. INTRODUCTION

Ecological risk assessment is a method for estimating which stressors or disturbances cause adverse effects on the integrity of ecosystems. The assessment process begins with planning and problem formulation, proceeds through analysis of exposure and effects, and ends with risk characterization.

Planning is the process of identifying interested individuals, management goals, and resource constraints for completing an assessment. Problem formulation includes a description of the ecosystem and its resources, driving forces, and stressors. Assessment endpoints based on the ecological and management goals are selected. These are woven together in a conceptual model that outlines the elements of the analysis phase. This is followed by quantitative and qualitative analyses of simulated and observed measures of exposure and effect. The uncertainty and variability of all elements of the analysis are explained. In risk characterization, the lines of evidence, type and severity of effect, and likelihood of recovery are discussed. Conclusions and recommendations for further analysis are drawn from a comparison of the uncertainty in the risk estimates and the evidence supporting the likelihood of ecological effect from exposure to the stressor(s).

This assessment is an analysis of environmental problems in the Middle Snake River, Idaho. The planning and problem formulation elements for this assessment were completed in 1996 (U.S. EPA, 1996). A synopsis of these elements is given in this report.

3. PLANNING

As a result of human activities spanning the past century, water quality and biological resource problems have developed in the Middle Snake River and its tributaries. The demands on the water resources have transformed this once free-flowing river segment to one with multiple impoundments, flow diversions, and increased physical, chemical, and microbiological pollutant loadings. Physical changes include significant alterations to rapids and pool areas. Biological changes include loss of native macroinvertebrate species, invasion and dominance of exotic species, an expansion of pollution-tolerant organisms, and excessive growth of macrophytes and algae.

The rapid rate of human population growth projected for the south Idaho region, as well as an increasing demand for energy and irrigation resources, commercialization of springs, and a burgeoning of dairy feedlots, will place additional burdens on an ecosystem that already has been substantially changed by human activity during this century. This ecological risk assessment was undertaken to address such concerns by analyzing the Middle Snake River's stressors and resulting ecological effects, and to stimulate broader public awareness and participation in decision making for reducing ecological risks. The ecological changes in this watershed have been observed by local, State, and Federal agencies; academic researchers; private organizations and businesses; recreational users; and individuals concerned about the loss of a species-rich ecosystem and cold-water fishery and degradation of water quality. The perspective with which local, State, and Federal planning agencies; scientists; and the general public view this watershed is changing as the community becomes more aware of how activities in the watershed impact the ecology of the river.

The development of a comprehensive watershed management plan involves close coordination of government, public, and private interests. Several working groups were formed to address both regulatory and nonregulatory issues. The agencies and organizations that have been identified as active in decision-making and management activities for the Middle Snake River include Federal, State, county, and private organizations; academic researchers; and interested citizens.

During the preliminary development of this analysis a variety of activities were undertaken to identify those interested in the area. A number of planning efforts were initiated by county officials (Mid-Snake River Planning Group) and the State of Idaho, Department of Environmental Quality Watershed Steering and Technical Committees. Most of the planning efforts were directed toward restoration of the cold-water biota and reduction of aquatic plant biomass in the Middle Snake River. A detailed list of the interested groups from 1987 to 1995 is presented in Appendix A. These original working groups have evolved into a Middle Snake

River Watershed Council.

Since 1969, several programs have been implemented to improve water quality in the Snake River Basin. The activities have included the advancement of best available technology at the municipal sewage treatment plant, regulation of waste handling at cattle feedlots, the initiation of best management practices on agricultural land through both State and Federal programs, permits for the aquaculture industry, and total maximum daily load limits for phosphorus.

The management goals for this watershed are associated with, and largely driven by, the specific requirements of State and Federal environmental legislation and the development of comprehensive land-use plans at the county level. These goals are:

- Attainment of State water quality standards;
- Establishment of total maximum daily loadings for water-quality-limited segments of the river;
- Sustained economic activity in the region;
- Water for hydropower and irrigation;
- Recovery of endangered species; and
- Recreational uses.

The goals for the risk analysis are determined by the state of our knowledge of the ecosystem and our ability to develop simulation models for the flow of energy, materials, and information between ecosystem compartments.

The goals of this risk analysis are to:

- Provide scientific information to address management goals;
- Develop an ecosystem perspective for environmental planning that can be used in other river basins throughout the region;
- Increase the knowledge of the structure and function of the Middle Snake River ecosystem; and,
- Expand the scope of simulation methods to include more complex compartments in the ecosystem.

In addition to advancing the science of risk analysis, this assessment is also undertaken to ensure that the public and special-interest users, government agencies, and scientists understand the ecological damage and that they develop a sense of partnership in reaching solutions for the recovery and protection of the Middle Snake River ecosystem. Too often, when such groups act in isolation, problems remain unresolved and each group becomes entrenched in its own rhetoric and territoriality. A consensus-building method of reaching shared solutions is inherently slow but fundamentally democratic. Recognizing deadlines, limited resources, and the continued decline of the habitat, it is important that progress be apparent.

The approach used to understand the interaction of sources, stressors, and resources on the Middle Snake River includes: (1) field studies and experiments to increase our understanding of the Middle Snake River ecosystem, (2) characterization of ecological risk using mathematical modeling methods, and (3) qualitative evaluation of biological changes. Ultimately, this risk analysis will be used to develop comprehensive management plans through the cooperative efforts of local, State, and Federal agencies; academic researchers; and an informed public. The analysis must reflect the interests of the interested parties. These measures alone cannot return the Middle Snake River to its original state, but they can provide a better environment for the natural heritage resources that have survived. Furthermore, if this approach is successful, the Snake River can provide an example for environmental stewardship in other river basins.

4. PROBLEM FORMULATION

In problem formulation the available information on stressors, ecological resources potentially at risk, and ecological effects is used to (1) identify the ecological resources (assessment endpoints) that will be the focus of the risk assessment, (2) develop conceptual models of how these resources may be affected by stressors, and (3) develop a plan for the analysis. In this report, a brief overview of the physical, chemical, and social forces that affect the natural ecosystem of the Middle Snake River is followed by the conceptual model and description of analytical methods. The results of the analysis are presented in Sections 6, 7, 8, and 9.

4.1. METEOROLOGY

The climate of the region is semiarid, characterized by low annual rainfall (e.g., 26.4 cm/yr at Twin Falls), with moderately hot summers and cold winters.

Precipitation is fairly evenly distributed throughout the year. November through January are the wettest months; July and August the driest. Precipitation during 1988-1993 was low, followed by a dramatic rise in 1996. The magnitude of precipitation is important because snow melt is a primary source of water in the Middle Snake watershed.

The average temperature from 1928 to 1989 was 10°C. The variation in temperature is also reflected in water temperatures.

Air temperature, relative humidity or dewpoint, cloud cover, wind speed, and atmospheric pressure are required inputs to the model for estimating the heat budget and the amount of solar energy available for heat transfer for primary productivity. For the purposes of this assessment, the Middle Snake River was divided into two meteorological provinces: one from Milner Dam to Upper Salmon Falls Dam, the other from Upper Salmon Falls Dam to King Hill.

4.2. GEOLOGY

The Snake River is the tenth longest river in the United States, extending 1,667 km from its origins in western Wyoming to its union with the Columbia River at Pasco, Washington. Along the way, it undergoes an elevation drop of about 2,895 meters. Its watershed (Figure 4-1) encompasses an area of approximately 267,000 km^2 in the States of Idaho, Oregon, Wyoming, Nevada, Utah, and Washington.

For the risk analysis, it was useful to characterize the length scales for the river in terms of geomorphologic, hydrologic, and cultural features. These segments are described in Table 4-1.

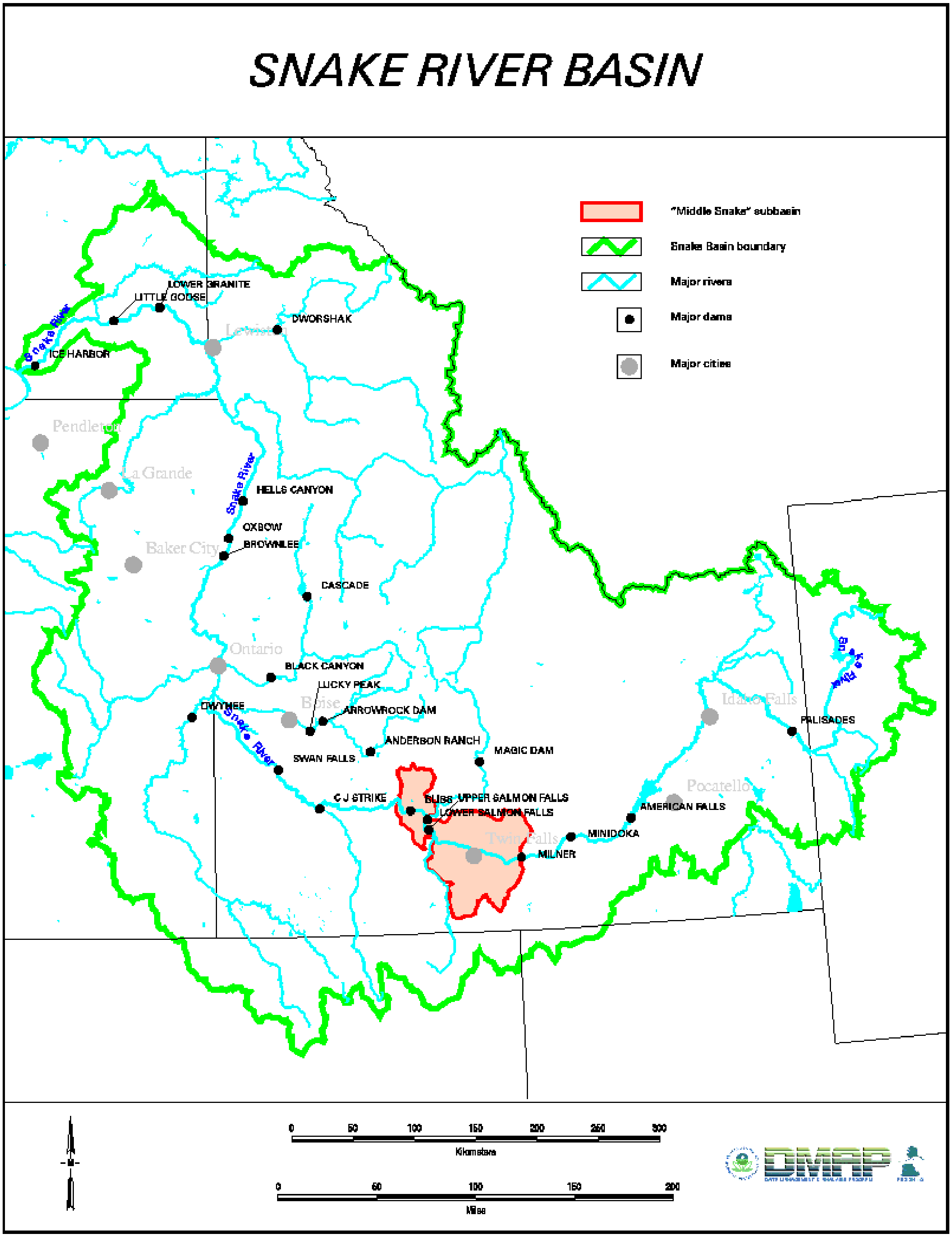

Figure 4-1. Snake River Basin.

Table 4-1. Hydrologic, geomorphologic, and cultural features of segments of the Middle Snake River between Milner Snake River between Milner Dam and King Hill

No.	Segment name	Initial Rkm[a] (RM)	Hydrology	Geomorphology	Cultural
1	Milner Dam to Twin Falls Reservoir	1,029.8 (640.0)	Dewatered during the irrigation season. Seepage and minor return flows on the south side. North side surface return flows.	The river has incised a moderate- to high-gradient canyon in basaltic and sedimentary rocks. Major rapids at Cauldron Linn (Rkm 1,016, RM 631.5).	Downstream from Milner Dam there are no hydraulic modifications of the river in this segment.
2	Twin Falls Reservoir to Shoshone Falls	996 (619.0)	Vinyard Lake and Twin Falls Coulee plus subsurface inflows from both north side and south side. Inflow from Devil's Washbowl, Dierke's Lake, and north side seeps. No north side surface return.	Snake Canyon widens at upper end. Two major falls: Twin Falls (approximately 150 feet high) and Shoshone Falls (approximately 200 feet high).	Hydroelectric project reservoirs with limited storage capacity.
3	Shoshone Falls To RM 609	989 (614.9)	East and Main Perrine Coulee return flows. No north side return flows, but some subsurface return flow.	Moderate gradient in basaltic rock with rapids at Pillar Falls (Rkm 986.3, RM 613.1). Canyon widens with talus and alluvium and unconsolidated pebbles, gravels, and boulders.	Major irrigation return flows and fish hatchery.
4	RM 609 to Rock Creek	979.9 (609.0)	Warm Creek, north side, south subsurface returns and some south side surface return flow.	High-gradient rapids in basalt with deep pool at the lower end of rapids.	City of Twin Falls STP discharge.
5	Rock Creek to Crystal Springs	976 (606.5)	North and south side subsurface and surface return flows.	Low gradient in basalt with little change in hydraulic section.	Major irrigation return flow.
6	Crystal Spring to Boulder Rapids	967 (601.0)	Niagara and Crystal springs, Cedar Draw inflow.	Generally low gradient with some minor rapids. Alluvial deposits of unconsolidated pebbles, cobbles, and boulders in basaltic sand.	Major irrigation return flows and fish hatcheries.

Table 4-1. Hydrologic, geomorphologic, and cultural features of segments of the Middle Snake River between Milner Dam and King Hill (continued)

No.	Segment name	Initial Rkm[a] (RM)	Hydrology	Geomorphology	Cultural
7	Boulder Rapids to Kanaka Rapids	960.6 (597.1)	Minimal inflow.	Generally low gradient with three major rapids. Alluvial deposits of unconsolidated pebbles, cobbles, and boulders in basaltic sand.	
8	Kanaka Rapids to Gridley Bridge	956.9 (594.7)	Mud Creek, Deep Creek, Clear Springs, Briggs Springs, Banbury Springs, Box Canyon Springs. South side irrigation return flows.	Low gradient, meandering river. Alluvial deposits of unconsolidated pebbles, cobbles, and boulders in basaltic sand grading to unconsolidated clay, silt, and sand.	Major irrigation return flows and fish hatcheries.
9	Gridley Bridge to Upper Salmon Falls Dam	937 (582.4)	Thousand Springs, Riley Creek.	Impounded river. Alluvial deposits of unconsolidated clay, silt, and sand.	Hydroelectric project reservoir with limited storage capacity.
10	Upper Salmon Falls to Lower Salmon Falls	934 (580.5)	Irrigation return flows, Billingsley Creek.	Impounded river. Alluvial deposits of unconsolidated clay, silt, and sand.	Hydroelectric project reservoir with limited storage capacity.
11	Lower Salmon Falls to Bliss Bridge	921 (572.6)	Malad River, springs.	Moderate gradient, freely flowing river in alluvial deposits of unconsolidated pebbles, cobbles, and boulders in basaltic sand.	
12	Bliss Bridge to Bliss Dam	910.2 (565.7)	Minimal inflow.	Impounded river. Some alluvial deposits of unconsolidated pebbles, cobbles, and boulders in some lacustrine deposits of clay.	Hydroelectric project reservoir with limited storage capacity.
13	Bliss Dam to King Hill	902.8 (559.9)	Some diversion for irrigation, north side irrigation return, Clover Creek.	Low-gradient river in alluvial deposits of unconsolidated pebbles, cobbles, and boulders and, in some places, basalt flows.	

[a]Rkm - river kilometer; RM - river mile.

The Snake River Plain comprises approximately 41,000 km^2 of the Snake River Basin in southern Idaho. The basin is subdivided into two geographic units: the eastern plain and the western plain. The boundary between eastern and western plains is near King Hill, Idaho.

The river reach of concern, hereafter referred to as the Middle Snake River, lies entirely within the eastern unit of the Snake River Plain. The study reach extends from Milner Dam (river kilometer [Rkm] 1,028; river mile [RM] 640) to King Hill (Rkm 877.6; RM 546.4).

The contributing watershed includes 22,326 km^2 (Figure 4-2) of land below the Milner Dam and adjacent to the study reach. Figure 4-3 shows a schematic diagram of the Middle Snake River, including the locations of all dams, tributaries, irrigation returns, and water withdrawals.

Land surface elevation ranges from 1,260.3 meters mean sea level (MSL) at Milner Dam to 762 meters MSL at King Hill. The predominant feature of the western part of the Snake River Basin, through which the Middle Snake River flows, is the relatively flat Snake River Plain, a structural downwarp filled with quaternary basaltic lava flows and bounded by interbedded sedimentary deposits (Clark, 1994). The geologic units include Pleistocene and older basaltic lava flows, pillow lavas (formed by lava flowing into water), alluvial deposits, and lake deposits from ancient lakes. The eastern plain is underlain by a thick sequence of volcanic rocks that store and yield large volumes of water, comprising the largest and most productive aquifer (Snake River Plain Aquifer) in the Northwest. The Snake River incises the aquifer just upstream of Twin Falls, near Kimberly. More than 80% of the groundwater emerges in the Thousand Springs area, breaking through hundreds of fissures or cracks in the basalt layers of the canyon walls (Travis and Waite, 1964).

The Snake River Canyon was scoured by overflow from the ancient Lake Bonneville during the Pleistocene, approximately 13,500 to 15,000 years ago. The flood waters deposited sandbars and gravel with boulders more than 3 meters in diameter. Many rapids and waterfalls are formed by these boulders. Below Milner Dam, the Snake River enters a deep (20-90 m) canyon cut through basalt and overlying sedimentary deposits and continues for 150 kilometers to King Hill. The river is incised in a steep-sided basalt canyon of about 91 to 122 meters depth through the reach.

Four major waterfalls over basalt ledges occur in the Middle Snake reach: (1) Star Falls at 8 meters, (2) Twin Falls at 34 meters, (3) Shoshone Falls at 65 meters (Figure 4-4), and (4) Auger Falls, a cascade that drops 12 meters. Average stream gradient is 3.4 meters/ km (0.33%) from Milner Dam to King Hill. Downstream of Twin Falls, Idaho, the Snake River canyon widens into small areas of bottomland and terraces. The largest of these areas is the Hagerman Valley, approximately 10 km long and 2 to 6 km wide.

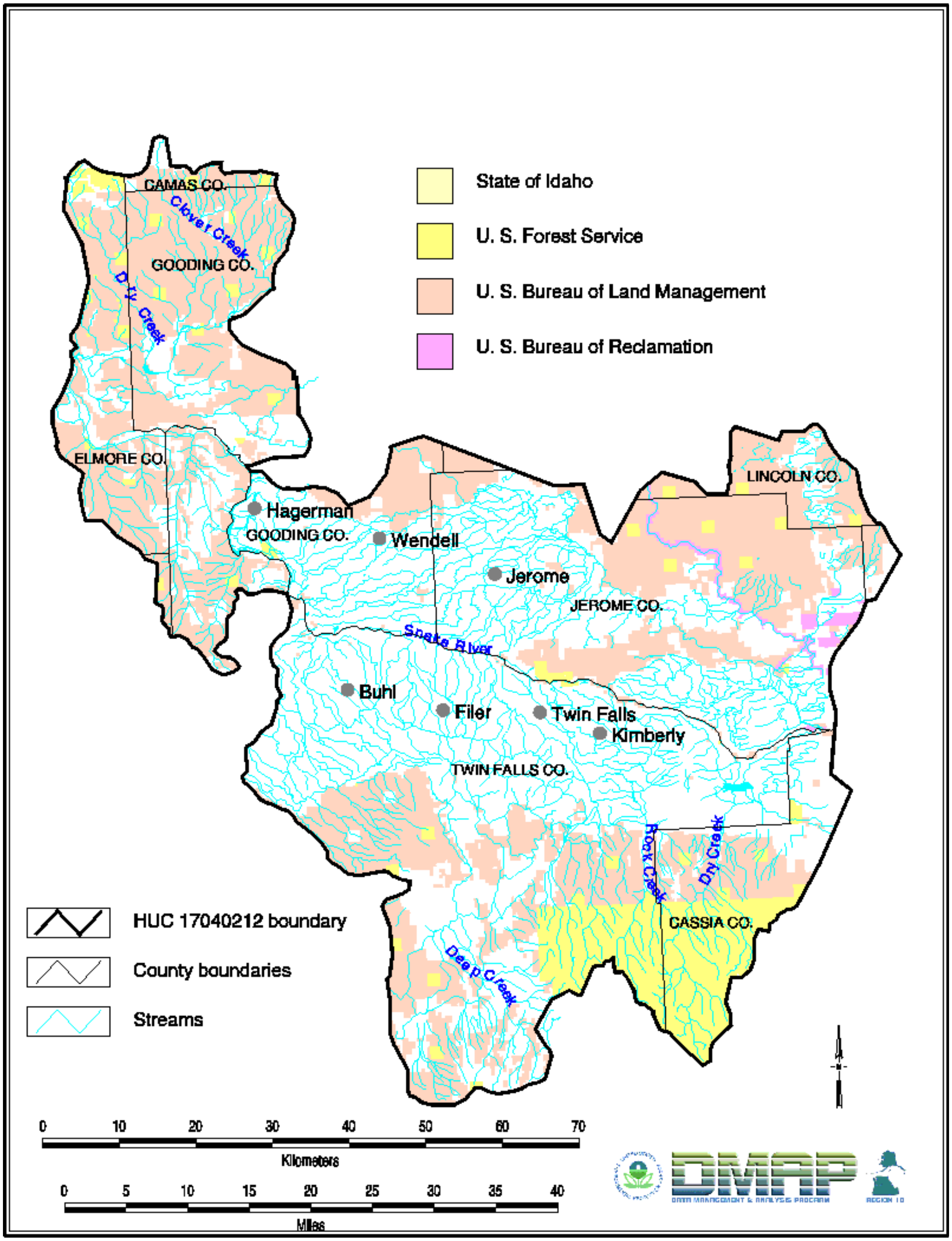

Figure 4-2. Hydrologic unit for the Middle Snake River.

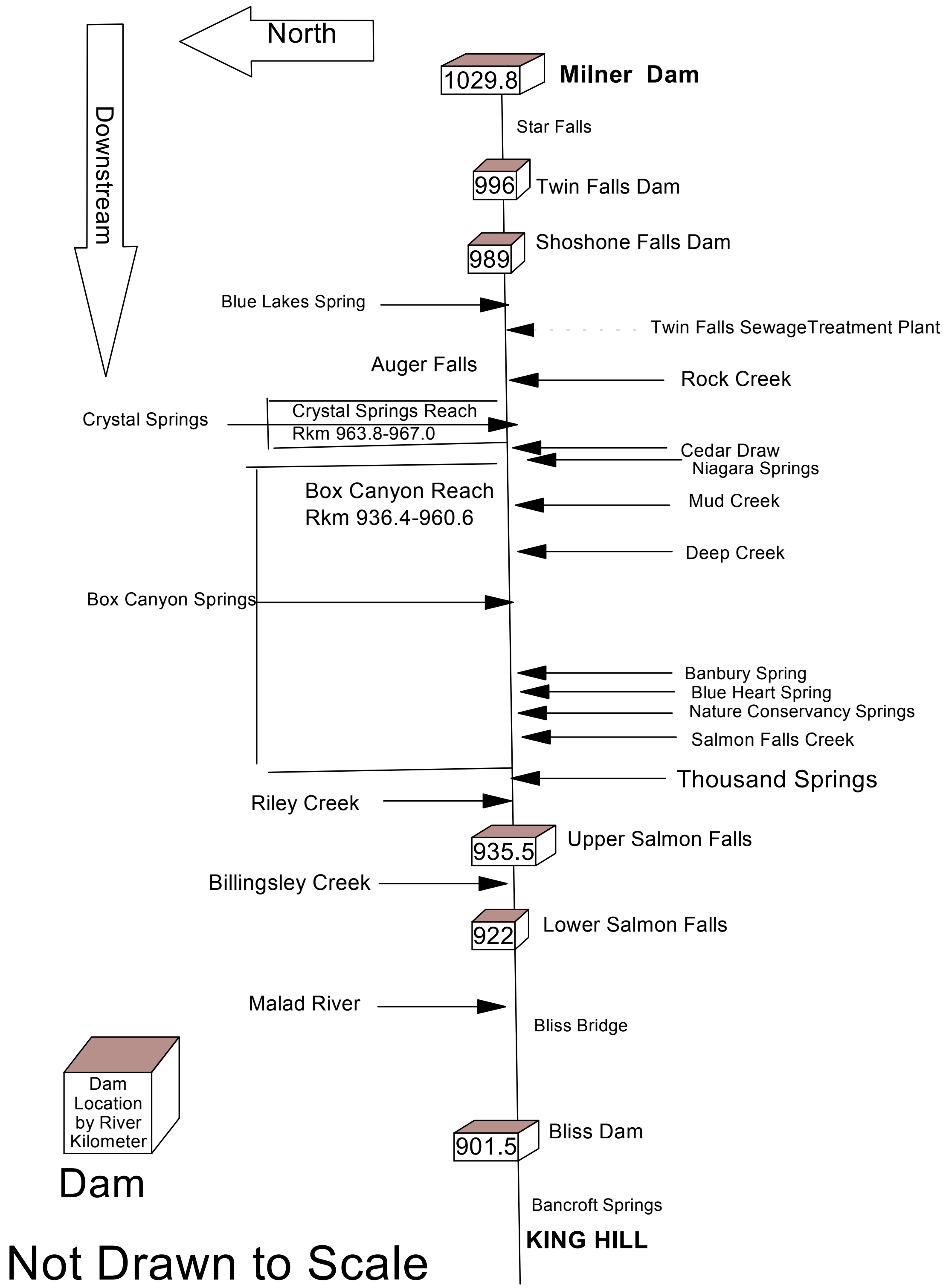

Figure 4-3. Schematic of the Middle Snake River from Milner Dam to King Hill, showing major tributaries, springs, dams, and point sources.

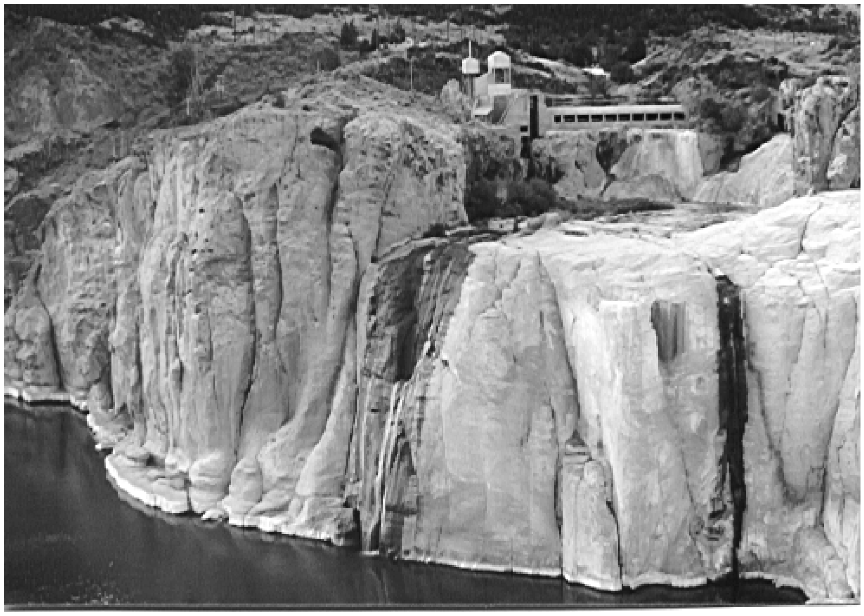

Figure 4-4. Shoshone Falls during low-flow conditions.

4.3. HYDROLOGY

The alteration of the natural hydrologic regime in the Middle and Upper Snake River began with the construction of the Swan Falls Dam in 1901 and continued with the construction of Milner Dam (1905), Minidoka Dam (1906), Lower Salmon Falls Dam (1910), Jackson Lake Dam (Wyoming - 1911), American Falls Dam (1927), Upper Salmon Falls Dam (1937), and finally Bliss Dam in 1949. Because of the management of this system as well as the geology, the hydrology of this segment of the Snake River is complex. Water sources are snow melt and groundwater recharge via springs. The Snake River above Milner Dam has an average annual flow of about 6×10^9 m^3/year (212×10^9 cfs/yr). Until recently the entire river was diverted at Milner Dam for irrigation during low-flow years from April to October. In 1992, an operating license issued by the Federal Energy Regulatory Commission to the Idaho Power Co. required that Milner Reservoir be kept full and a target flow of 6 m^3/s be released, if available. Mean annual water flow at Milner Dam is 92.3 m^3/s (3,259 cfs) (1910-1990 record). *Average* flow at Milner Dam disguises the fact that low flows (in the mid- and late irrigation season) may approach zero as a result of water diversion from the channel. Over the 1980-89 period, flows from Milner Dam were less than 2.8 m^3/s (99 cfs) 10% of the time (Clark, 1994), usually during the summer irrigation months. An inspection of the U.S. Geological Survey (USGS) historical streamflow daily graphs from 1970 to 1997 shows that the annual low flow for the King Hill station (Figure 4-5) may occur in any month except November and December. However, low flows usually occur from July to September. In low- to average-flow years, flows decline

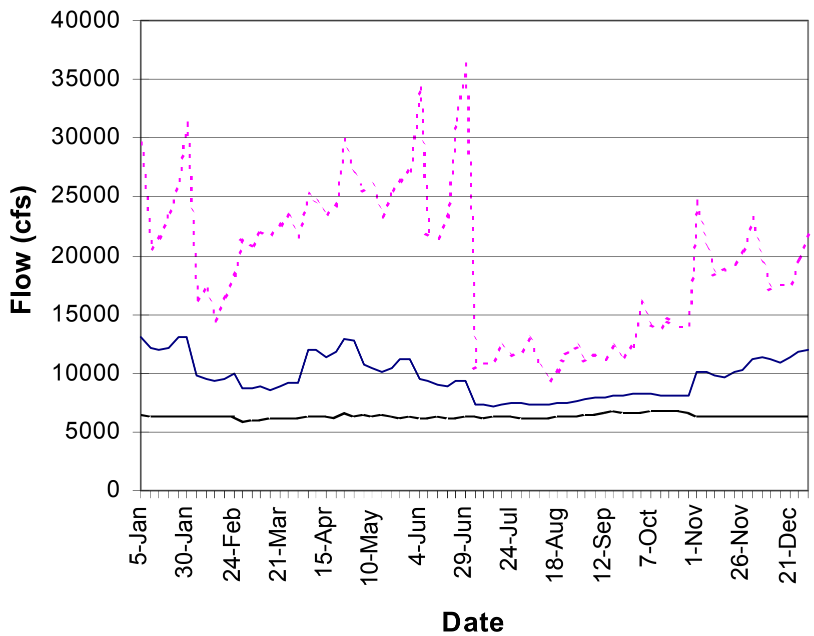

Figure 4-5. Flow (cfs) in the Middle Snake River at Rkm 893 (RM555), mean + or - 2 SD. 1cfs = 0.02832 cms.

through the winter to early summer unless a higher flow year such as 1993 causes spill from Milner (Myers et al., 1995). With low precipitation during 1988-1993, the flows were extremely low. However, in 1996 there was a dramatic rise in precipitation and concomitant snow melt with nearly 1,670 m^3/s (58,968 cfs) flowing over Shoshone Falls at the height of runoff.

Downstream from Milner Dam, flows increase substantially (average flow is 3×10^9 m^3/year or 106 cfs) because of tributaries, groundwater discharge, and irrigation returns. There are eight major tributaries to the Middle Snake River (Figure 4-2): East Perrine Coulee, Rock Creek, Cedar Draw Creek, Mud Creek, Deep Creek, Salmon Falls Creek, Billingsley Creek, and the Malad River. The total contribution of these tributaries averages 48 m^3/s (1,695 cfs), with the major contribution from the Malad River.

In his water budget analysis for the entire Snake River Plain during 1980, Kjelstrom (1992) found that groundwater contributed 146 m^3/sec (5,155 cfs) of flow to the Middle Snake River segment. This represents more than 50% of the average annual flow at Lower Salmon Falls. Kjelstrom (1992) reports, however, that groundwater discharge to the Snake River has varied as recharge conditions have changed. From 1902 to the early 1950s, groundwater discharge to the Middle Snake River segment increased because of recharge from flood irrigation on the north side of the Snake River. In the 1950s, the estimated average annual groundwater flow to the Middle Snake exceeded an estimated 190 m^3/s (6,709 cfs). Since that time, flows declined until 1992 because of drought conditions in the basin and increases in groundwater pumpage from the Snake Plain aquifer, with an accompanying shift from flood to sprinkler irrigation (Kjelstrom, 1992). In 1996, the increased precipitation should have increased the groundwater flow as well as the surface water flows.

Mundorff et al. (1964) found that the total gain from the aquifer to the Snake River between Milner Dam and King Hill is equal to about two-thirds of the discharge measured at the USGS gauge at King Hill. The major springs are Devil's Washbowl, Devil's Corral, Warm Creek, Crystal Springs, Niagara Springs, Clear Springs, Briggs Springs, Box Canyon Springs, Sand Springs, Thousand Springs, Riley Creek, and Malad Springs.

North-side springs to the river from about Rkm 957.5 (RM 595) upstream are supplied primarily from local surface water recharge in agricultural areas between Minidoka and Twin Falls, whereas springs below Rkm 957.5 (RM 595) are derived primarily from regional groundwater, mainly intermontane basin stream recharge (Clark, 1997a). Clark and Ott (1996) estimated that the upstream springs received more than 90% of their flow from irrigation recharge, whereas the downstream springs received less than 20% of their flow from irrigation recharge. When most of the river is diverted during the irrigation season, the springs are the primary source of river flow.

4.4. DEMOGRAPHICS AND LAND USE

The political boundary of the study area includes six Idaho counties (Twin Falls, Jerome, Gooding, Lincoln, Blaine, Camas) and portions of Minidoka, Cassia, Owyhee, and Elmore. This area is commonly referred to as the Magic Valley. About 136,831 people (11% of the population of the State of Idaho) live along the Snake River. The five largest municipalities in the Middle Snake study area are Twin Falls (27,951), Burley (8,984), Jerome (6,529), Rupert (5,455), and Hailey (3,687) (Figure 4-2). The remaining population lives in unincorporated areas.

About 26% of the land is privately owned, 70% is Federal land, and the remaining 4% is State land. The primary land use by surface area is irrigated agriculture (23%) and grazing (56.4%) (Figure 4-6). Forest and urban land make up less than 7% of the total land use.

Most of the land adjacent to the Middle Snake River is used for agriculture, roads, golf courses, small cattle operations, private homes, boat docking facilities, and fish hatcheries. Recreation activities include fishing, boating, and swimming in some limited areas.

Key industries in the area are agriculture, livestock production, and aquaculture. The primary crops are potatoes, sugar beets, and barley. Most of the livestock production is dairy cows. Seventy-six percent of the trout produced in the Nation come from Idaho.

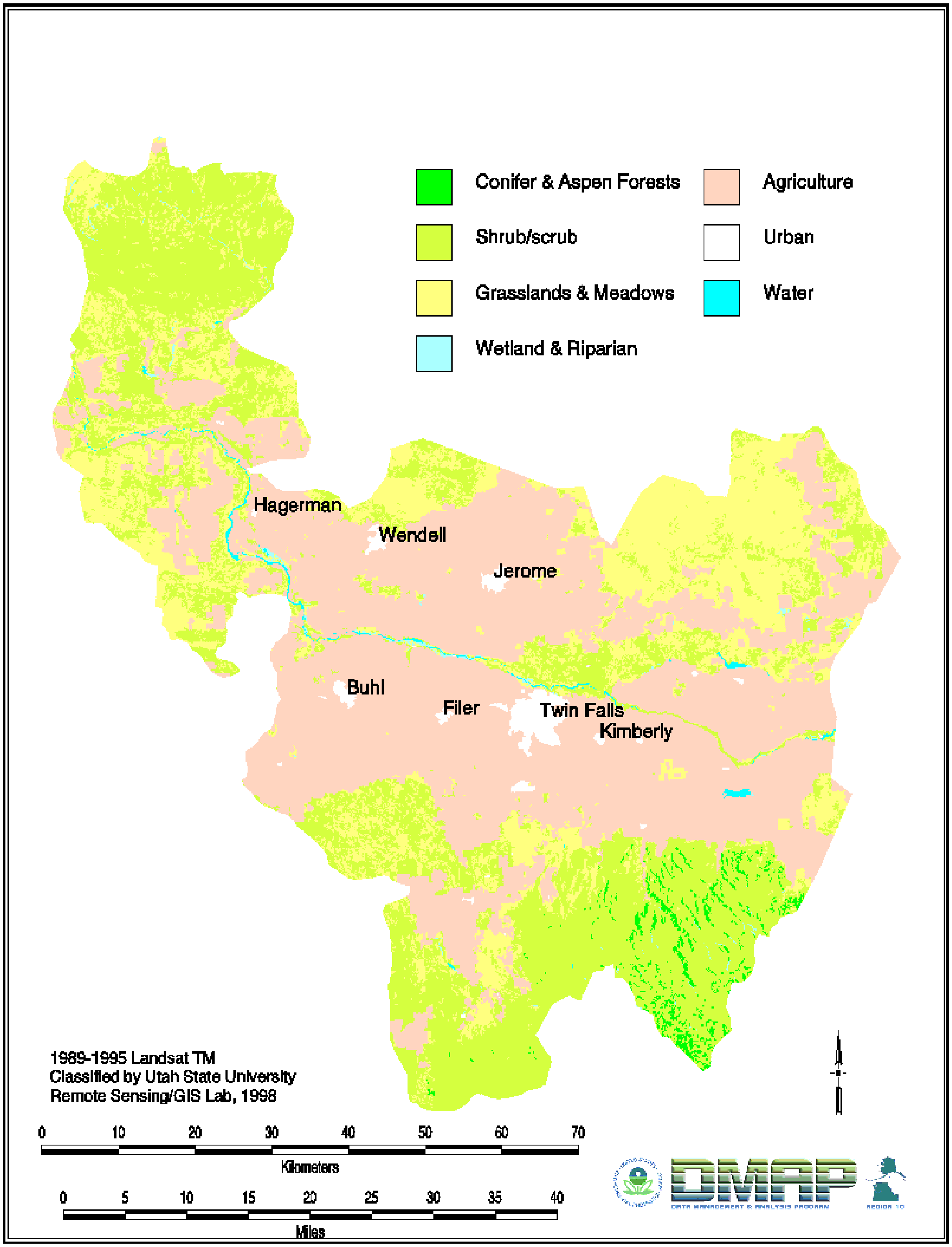

Figure 4-6. Generalized land cover for the Middle Snake River Basin.

Table 4-2. Primary land-use activities in the Middle Snake River between Milner Dam and King Hill, Idaho (from Bowler et al., 1993)

Point sources	Quantities
Combined animal feeding operations	600 dairies and feedlots
Aquaculture	80 facilities
Publicly owned treatment works	Twin Falls
Nonpoint sources	**Quantities**
Irrigated agriculture and cattle grazing	227,000 hectares irrigated from the Snake River; 150,000 hectares irrigated from the Snake River aquifer; Return flow from 13 streams and >50 surface drains
Impoundments, diversions, and hydroelectric facilities	5 existing on mainstem; 7 proposed on mainstem; many on tributaries

For the purposes of this analysis, land-use activities (Table 4-2) have been divided into three categories: (1) point sources of pollutants, (2) nonpoint sources of pollutants, and (3) structural alterations.

Sources upstream of Milner Dam are not included as individual releases in this assessment. The total load from Milner Reservoir is included as a single discharge point in the analysis.

4.5. FISH POPULATIONS

Before 1900, the Snake River was the most important drainage in the Columbia River system for the production of anadromous fishes. Prior to the development of hydropower on the Snake River, the Middle Snake sustained a variety of anadromous fish species that migrated as far upstream as Shoshone Falls. This included fall and summer chinook salmon (*O. tshawytscha*), steelhead trout (*O. mykiss*), and Pacific lamprey (*Lampetra tridentata)*. The anadromous salmonids were first severely impacted by the construction of Swan Falls Dam in 1901 without adequate fish passage facilities. The final major hydroelectric events resulting in the termination of migrant fish stocks were the sequential closures of the Bliss Dam (1949), C. J. Strike Dam (1952), and ultimately Brownlee Dam (1960). All these dams form impassable barriers to the upstream movement of any anadromous species and sturgeon because they were constructed without any fish passage facilities.

The completion of these facilities terminated lamprey, salmon, and steelhead migration into the Middle Snake area (Smith, 1978; Bowler, 1992). The remaining downstream Snake River stocks of fall and spring/summer chinook salmon have been listed as threatened or endangered (USFWS, 1995). Although a number of impoundments presently block the migration of anadromous salmonids, a number of resident cold-water species, including trout and sturgeon, have survived in the river and tributaries.

Currently, there are approximately 24 native fish species below Shoshone Falls in the subbasin and 14 above the falls (Appendix B). White sturgeon (*Acipenser transmontanus*), rainbow and steelhead trout (*Oncorhynchus mykiss*), and mountain whitefish *(Prosopium williamsoni*) are native to the Middle Snake River.

The majority of the remaining fish in the Middle Snake are eutrophic-tolerant species, such as some catostomids (suckers), northern pike minnow (*Ptychocheilus oregonensis*), the non-native European carp, and various other cyprinids.

4.6. BENTHIC MACROINVERTEBRATES

The historic diversity of native molluscs in the river was high at 42 species, including 27 species of snails in 7 families and 15 species of clams in 3 families (Frest and Bowler, 1993). In the past, the river supported a diverse cold-water macroinvertebrate fauna (in addition to the molluscs and crustacea), including numerous Ephemeroptera, Plecoptera, and Trichoptera. Currently the benthic community (see Appendix B) is dominated by a few taxa indicative of degraded conditions (Dey and Minshall, 1992). These taxa include *Potamopyrgus antipodarum* (Gray, 1843), Chironomidae, Oligochaeta, and Hyallela. The exotic New Zealand mudsnail *(P. antipodarum*) is now the dominant mollusc as well as the dominant benthic macroinvertebrate.

The hydrobiid *P. antipodarum* is native to New Zealand. It was first recorded in the Middle Snake River by Taylor (1987). It has been observed attached to algae, macrophytes, and rocky boulder habitats (Bowler, 1991). By 1989, *P. antipodarum* dominated the benthic macroinvertebrate habitats in the Middle Snake River (Bowler, 1991). *P. antipodarum* dominates the preferred habitat of other hydrobiid snails and physically covers their egg-laying sites. The species also crowds out other species as the density of its population increases to 600,000 individuals /m^2 (Bowler and Frest, 1992).

The large freshwater clam *Margaritifera falcata*, once a food staple for Native Americans along the river, is now virtually eliminated from the Middle Snake. The decline may be due to sedimentation, loss of rapids (Vannote and Minshall, 1982), or a significant reduction in the juvenile salmonid population, which serves as a host for the parasitic larval stage of this organism. Although *M. falcata* is common in the Blackfoot River and elsewhere in the Upper Snake, the species has been replaced by the smaller pelecypod *Gonidea angulata* (Bowler and Frest, 1992) in the Middle Snake River.

The following eight species are listed under the Endangered Species Preservation Act as threatened, endangered, or species of concern:

Threatened:

(1) the Bliss Rapids snail, *Taylorconcha serpenticola* (Hershler et al., 1994)

Endangered:

(2) the Utah valvata snail, *Valvata utahensis* (Call, 1884)

(3) the Snake River physid snail, *Physa natricina* (Taylor, 1988)

(4) the Idaho springsnail, *Pyrgulopsis idahoensis* (Pilsbry, 1933) (also known as the Homedale Creek springsnail)

(5) the Banbury Springs limpet (undescribed *Lanx* sp.)

Species of concern:

(6) the California floater, *Anodonta californiensis*

(7) the giant Columbia River limpet, *Fisherola nuttalli (*Haldeman, 1841)

(8) the Columbia River spire snail, *Fluminicola columbiana* auct.

The Banbury Springs limpet, Snake River *Physa*, the Bliss Rapids snail, and the Idaho springsnail are found nowhere else outside of the Middle Snake River. They are endemic to the ancient Lake Idaho, which once covered most of the area during the Pliocene.

4.7. AQUATIC PLANT COMMUNITIES

Aquatic plant composition and densities, as well as patterns of mixed vascular, epiphyte, and periphyton interactions, are highly variable through a rapidly changing series of habitats of the Middle Snake River.

Aquatic vascular plants through this reach are generally dominated by *Ceratophyllum demersum, Potamogeton pectinatus*, and *P. crispus* in reaches of significant attached plant growth (Falter and Carlson, 1994). *Ceratophyllum demersum* and *P. pectinatus* are generally associated with well-buffered, nutrient-rich waters (Filbin and Barko, 1985; Best and Mantai, 1978). Subdominants are *P. foliosus*, *Elodea nuttallii*, and *E. canadensis*. *Ceratophyllum* and *Elodea*, although vasculars, lack true roots and obtain most needed nutrients from the water column. They are therefore considered functional epiphytes along with filamentous algae. Other primary components of this epiphyton community are the filamentous green algae, *Cladophora* sp., *Hydrodictyon* sp., and *Enteromorpha*. There are many locations in the Middle Snake where epiphyton is the principal component of the total summer-fall macrophyte biomass (Falter and Carlson, 1994; Falter et al., 1995, Falter and Burris, 1996).

Blooms of planktonic (*Microcystis, Cyclotella, Ceratium*), periphytic, and epiphytic algae (*Cladophora, Hydrodyction*) occur continuously during the spring and summer in specific reaches of the Middle Snake. The total epiphytic algae and vascular macrophyte biomass may exceed 2,000 g/m^2 dry weight, with *Cladophora* averaging 50% of the plant biomass in summer months (Falter et al., 1995).

A detailed life history of the dominant attached aquatic plants is given in Appendix C.

4.8. ASSESSMENT ENDPOINTS

Assessment endpoints are explicit expressions of the actual ecological values that are to be protected (U.S. EPA, 1998). These endpoints form a basis for linkage to risk management activities in the watershed. The endpoints for this analysis were selected in 1996 (U.S. EPA Problem Formulation, 1996). They are:

- The reproduction and survival of three fish species:
  White sturgeon (*Acipenser transmontanus*), mountain whitefish, *(Prosopium williamsoni*), and rainbow trout *(Oncorhynchus mykiss)*.
- The reproduction, survival, and diversity of macroinvertebrates:
  Bliss Rapids snail (*Taylorconcha serponticola*), Utah valvata (*Valvata utahensis*), Snake River physa (*Physa natricina*), Idaho springsnail (*Pyrgulopsis idahoensis*), and Banbury Springs lanx (undescribed *Lanx* sp.).
- The growth of periphyton, macrophytes, and epiphytes:
  Potamogeton pectinatus, P. crispus, Ceratophyllum demersum, Elodea canadensis, Hydrodictyon, Cladophora, Spirogyra, and *Enteromorpha.*

The growth of periphyton, macrophytes, and epiphytes was selected as an assessment endpoint because their presence at an appropriate level is ecologically important for protecting cold-water fish and macroinvertebrates. Had this assessment started after publication of the Guidelines for Ecological Risk Assessment (U.S. EPA, 1998) growth of periphyton, macrophytes, and epiphytes may have served as a measure of effect for the other two assessment endpoints.

Representative species from three major trophic levels were chosen as endpoints in order to complete an ecosystem-level analysis. Each of these groups (fish, invertebrates, and plants) is an important link in the structure and function of this riverine ecosystem. Analysis of the factors controlling their functions (growth, reproduction, and survival) should provide evidence for the primary causes of the ecosystem changes.

In addition to being indicators of ecosystem structure, fish and macroinvertebrates were selected as assessment endpoints because they exhibit marked sensitivity to stressors, and changes in populations of these assemblages can be linked quantitatively to several environmental parameters (e.g., numeric criteria) to document the stressor and ecological response relationships.

Target fish species for this study were rainbow trout (*Oncorhynchus mykiss*), mountain whitefish (*Prosopium williamsoni*), and white sturgeon (*Acipenser transmontanus*); all are cold-water species of recreational importance in the Snake River. An assessment of the life stage requirements for these species will provide an overview of most freshwater habitats used by native fish species in the Middle Snake River. The macroinvertebrates were also selected because the populations are either threatened or endangered. The decline of native species indicates that they are sensitive to the changes that have occurred in the Middle Snake River. An analysis of the factors contributing to their decline is necessary in order to preserve the remaining numbers as well as promote recovery for the populations.

The high aquatic plant densities are indicators of ecological conditions (eutrophication) that are not conducive to the growth and survival of cold-water biota. The reduction of aquatic plant biomass is an essential step to the restoration of cold-water biota.

4.9. DECISION PATHWAY

The decision pathway (Figure 4-7) for this risk analysis begins with the description of the land-use activities that may result in harm to the riverine ecosystem. For those properties that are

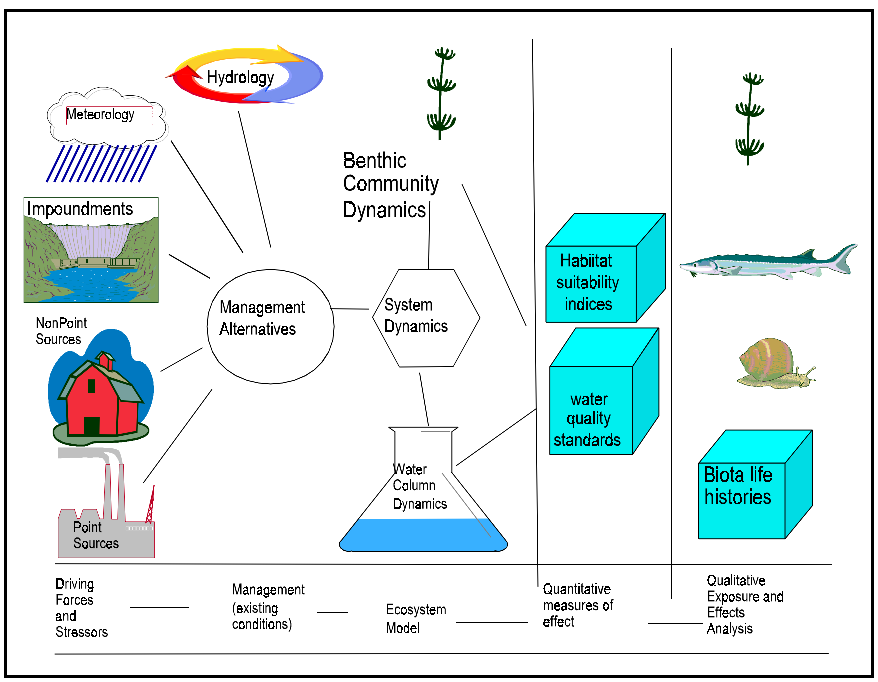

Figure 4-7. Decision pathway for analysis of ecological risk using simulation methods.

quantitative measures of stressors and ecological effects, simulation with mathematical models can be used to make quantitative estimates of ecological risk. Stressor characteristics are defined in terms of probability models for point source loadings, nonpoint source loadings, and meteorologic and hydrologic conditions. These characteristics are used as forcing functions for a mathematical model of the river ecosystem and to develop cumulative distribution functions for environmental factors such as dissolved oxygen, temperature, and macronutrients. The mathematical model developed by Yearsley (1991, modified in 1996) uses standard kinetics to simulate temperature, dissolved oxygen, nitrogen, phosphorus, and primary productivity for time scales of hours to decades, vertical length scales of meters, and horizontal length scales of meters to kilometers. Limitations in our understanding of ecosystem processes in the Middle Snake River are such that the model does not simulate all the variables that characterize the primary stressors described in the introduction. In particular, the model does not include those variables necessary to characterize sediment loading and habitat alteration associated with changes in the substrate.

The quantitative risk is estimated by comparing simulated measures of temperature, dissolved oxygen, phosphorus, and macrophyte biomass with quantitative measures of effect.

Measures of effect are quantitative estimates of the state of the ecosystem that can be related in some way to the values expressed by the assessment endpoints. For this analysis, the Idaho water quality standards and U.S. Fish and Wildlife habitat suitability indices are considered to be quantitative measures of effect.

Finally, the quantitative risk estimates are analyzed qualitatively using best professional judgment and field observations.

A detailed description of the simulation methods, results, and uncertainty analysis is presented in Appendix D of this report.

4.10. CONCEPTUAL MODEL

The conceptual model (Figure 4-8) for this assessment illustrates the land-use activities, stressors, ecological processes affected, and biological consequences of these process changes.

The hypothesis for this analysis is that flow and temperature alteration, sediment deposition and scouring, ammonia toxicity, decrease in dissolved oxygen, and nutrient loading are the principal stressors in this ecosystem. These stressors interact with the biota, causing a decline in native cold-water biota as a result of individual or synergistic influences.

The parameters identified as stressors (flow, temperature, ammonia, dissolved oxygen, sediments, and nutrients) are driving forces in natural ecosystems. They are the physical and chemical characteristics that define the structure and function of ecosystems. It is only when these parameters exceed biological tolerance limits that they become stressful or harmful to

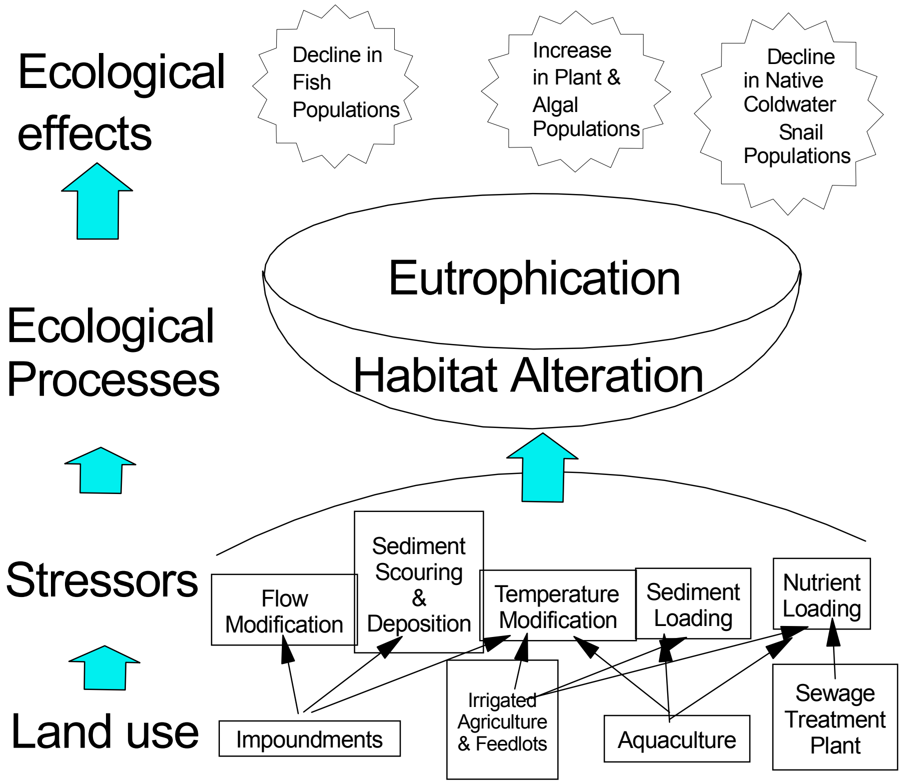

Figure 4-8. Conceptual model for the Middle Snake River Risk Assessment.

aquatic life. The tolerance limits are generally equal to the natural levels that have defined the ecological boundaries for which most native species have adapted. Excursions above and below these boundaries or tolerance levels can be stressful. These tolerance levels depend on life stage and vulnerability of the organisms at risk as well as the likelihood of exposure or contact with the stressful environment.

4.11. LAND-USE ACTIVITIES THAT ALTER ECOSYSTEMS

Land-use activities can affect ecosystem structure and function through point and nonpoint source release of pollutants (thermal, chemical, physical) and physical disturbance. Sources of ecological stressors identified in the conceptual model as point sources include the Twin Falls Sewage Treatment Plant, confined animal feeding operations, and aquaculture facilities. These facilities release nutrients and sediments through discharge canals and pipes directly into the river. The nonpoint sources include irrigated agriculture and cattle grazing. These activities result in releases to groundwater and surface water through leaching and runoff. Finally, impoundments cause physical changes to the river ecosystem that can be harmful to native biota.

4.11.1. Twin Falls Sewage Treatment Plant

There is only one sewage treatment plant in the study area. The Twin Falls Sewage Treatment Plant discharges sewage after secondary treatment. The plant uses an activated sludge system designed to treat 7.8 million gallons per day (mgd) of wastewater. The facility consists of the following unit operations: bar screens, grit removal, primary clarification, activated biofilter tower, intermediate clarification, activated sludge, secondary clarification, and ultraviolet disinfection. A city-owned anaerobic digester was recently added between Lamb-Weston (formerly Universal Frozen Foods) and the treatment plant to digest potato solids before they reach the plant. The Twin Falls facility discharges nutrients, ammonia, settleable solids, total suspended solids, and organic matter.

4.11.2. Confined Animal Feeding Operations

Confined feeding operations are required to contain all wastewater and are allowed to discharge only during extreme rain events (once in 25-year 24-hour storms). Unfortunately, such events do occur, and during these events or because of accidental or illegal discharges, nutrients, pathogens, and sediments reach the river through surface runoff and via groundwater contamination. The Middle Snake River area is very popular for dairy operations because of the climate and close proximity to cheese factories. Dairies and feedlots dispose of their liquid and solid wastes through land application, primarily on cropland. There are more cattle in the Magic Valley than in the entire rest of the State.

4.11.3. Aquaculture

There are 80 private and State-owned aquaculture facilities that have been operating for more than 30 years in the Middle Snake River. They are required to obtain Federal National Pollutant Discharge Elimination System (NPDES) permits. More than 20 additional facilities have applied for permits to discharge.

These facilities operate earth and concrete raceways in series or in parallel on a continuous or batch basis. These include both cold-water facilities, which raise trout, steelhead, salmon, and sturgeon; and warm-water facilities, which raise catfish, tilapia, and carp. The annual production of these facilities ranges from 9,072 kilograms to more than 453,600 kilograms. They supply approximately 80% of the trout consumed in restaurants in the United States.

Discharges from aquaculture operations typically contain organic and inorganic solids, chemicals used in prevention and treatment of disease, and nutrients. Discharges could impact water quality in the receiving stream by adding ammonia, bacteria, dead fish, feces, residual disinfectants and disease-control drugs, settleable solids, thermal energy, and total suspended solids.

Several aquaculture facilities have associated fish processing facilities that butcher fish for market onsite. Production ranges from hundreds to tens of thousands of trout, catfish, or tilapia per day. Pollutant discharges from the fish processors consist of rinse and washdown water and entrained blood and gut remnants, measured in terms of biochemical oxygen demand, total suspended solids, settleable solid residues, nutrients, disinfectants, and pH.

Pollution reduction by Idaho's aquaculture industry began with the construction of settling ponds in the mid- to late 1970s. Effluent from raceways and rearing ponds would pass through these ponds slowly, allowing solids to settle before the facility discharge point (Aquaculture Watershed Reduction Plan for the Middle Snake River, 1997; e.g., Brown et al., 1974; Kendra, 1991; Westers, 1989). By 1984, a number of aquaculture facilities were experimenting with the use of screens to keep resident fish from congregating within 3 to 6 meters of the effluent weir in each raceway (JRB, 1984). These areas of the raceways became known by industry as quiescent zones. They were effective at settling solids in the raceway, allowing industry to meet the 5.0 mg/L total suspended solids limit on raceway discharges. Settled solids were removed either by mechanical or siphon vacuuming or by draining through opened stand pipes in the quiescent zone. Facilities were also experimenting with the effectiveness of solids removal using standpipe siphon hydraulics. Vacuumed or siphoned solids would be sent to off-line settling ponds for further treatment. Improved feed conversions, lower phosphorus feeds, and improvements in availability of phosphorus in feeds are believed to have reduced phosphorus discharges by the industry during the 1990s (Aquaculture Watershed Reduction Plan for the Middle Snake River, 1997).

A study of six fish farms discharging to Deep Creek, a tributary of the Middle Snake River, was completed in 1993 by the University of Idaho (Deep Creek Fish Farm Effluent Study, Collins and Brannon, 1994). Because of the quality of the source water (Deep Creek), these fish farms had a negative net contribution of suspended solids and nitrite-nitrate levels, but they had a positive net contribution of ammonia and phosphorus in their effluent. The study found that solids and dissolved nutrients can be reduced in settling areas below rearing ponds, at least in the low-fish-density ponds of this study. It is much more difficult to achieve settling in high-fish-density, high-flow raceways.

4.11.4. Irrigated Agriculture and Cattle Grazing

Agriculture is made possible by water withdrawal from the Snake River. Early settlers used water from the Snake River tributaries for irrigation. In the summer of 1903, the Twin Falls South Side Land and Water Company tract was opened to farmers (IDEQ, 1995) for irrigation of their crops. The Twin Falls North Side Land and Water Company was granted permission to construct canal systems and withdraw water from Milner Reservoir under the provisions of the Federal Carey Act in 1907.

Poor agricultural practices from crop production can result in increased sediment loading. The Soil Conservation Service's River Basin Reports of 1976, 1979, and 1981 identified substantial areas of serious erosion on surface-irrigated lands in the Upper Snake River basin. Gooding and Jerome Counties each had more than 20,000 hectares with erosion rates exceeding 1.8 metric tons/hectare/year, while Twin Falls County had between 2,000 and 20,000 hectares exceeding 1.8 metric tons/hectare/year.

Sediment loads increase dramatically with increased runoff flow rates from cropland (Carter, 1976). Greater rates of flow off the land into irrigation-return canals increase the amount of the sediment inputs into the streams and river. Irrigation return flows carry pesticides, fertilizers, and sediment loads to the river. Runoff from individual fields, especially those using furrow irrigation, carries sediment into drainage canals, which eventually reaches the river. Different crops yield different levels of sediment, e.g., sediment loss from alfalfa fields is fairly low whereas that from dry-bean production is fairly high.

Most of the smaller canals that flow over the precipitous canyon wall percolate through talus debris piles formed from rock falling off the canyon wall. Accumulated sediment and rock debris tend to remove some of the other pollutants associated with irrigation wastewaters in a fashion similar to wastewater treatment by land treatment systems. During heavy rains or after snow melt, the overflow into the river occurs with little or no percolation through debris piles. Most larger irrigation return flows are much more damaging to the river. Irrigation return flows at the Perrine Coulee hydroelectric facility (NPDES Draft Permit, 1998) are conveyed through a penstock to a hydroelectric turbine. Thus, the water bypasses the talus slope and is discharged directly to the river, creating a sediment-laden pollutant plume.

Although some farmers have incorporated low-till and other best management practices as part of their cultural practices, implementation of best management practices is not widespread in the region.

Agricultural practices also result in the release of nutrients into the groundwater and into surface waters of the watershed. Carter et al. (1971) estimated that 2,737 metric tons of nitrate were transported from the Twin Falls irrigation system into the Snake River. Sediment from Twin Falls was estimated at 2,377 metric tons/year (Brown et al., 1974).

4.11.5. Nutrient and Sediment Loading

The total industry loading estimates are presented in Table 4-3 (from IDEQ's Nutrient Management Plan, 1995). From this table, it is obvious that agriculture is the primary source of sediments and that springs are the primary source of nitrogen. The nitrogen load from springs is a result of leaching of wastes from agricultural, cattle grazing, and cattle feedlots.

Water chemistry data collected by the University of Idaho Agricultural Research Station, the Idaho Division of Environmental Quality (IDEQ), Clear Springs Food Inc., and the City of Twin Falls were used to estimate daily mass loadings for point sources in the study.

4.11.6. Impoundments

Impoundments that store and divert water for hydropower and irrigation result in flow modifications in the mainstream and tributaries. Summer-fall flows into this reach are controlled by several large upstream storage reservoirs (American Falls, Island Park, Palisades, and Ririe Reservoirs). There are five existing impoundments within the study area downstream from

Table 4-3. Estimated nutrient and sediment loadings for point, nonpoint, and background sources. This table is an excerpt from the State of Idaho Department of Environmental Quality Nutrient Management Plan for the Middle Snake River (from IDEQ, 1995). These estimates are based on weighted mean net discharge levels reported by the industry, which were averaged with estimates of net contributions estimated by Brockway and Robinson (1992). The result is an industrywide net contribution. These loads are based on an assumption of industrywide water usage of 85 m^3/s (IDEQ, 1995).

Sources	Sediments (TSS)[a] (kg/day)	Total phosphorus (kg/day)	Total nitrogen (kg/day)	References
Upstream	0.3 (251)	16 (282.5)	118 (282.6)	Brockway and Robison, 1992
Springs	0	359 (304)	27,713 (22,150.6)	Brockway (unpublished), MacMillan, 1992; Clark, 1994
Aquaculture	13,497	733	5,794	1991 DMRs[b]; Brockway (unpublished)
Twin Falls POTW	733	467		1991 and 1992 DMRs
Irrigated agriculture	157,873	276	7,097	Brockway and Robison, 1992
Other	42,876	228	2,336	Brockway and Robison, 1992

[a]Total suspended solids.

[b]Discharge monitoring reports from NPDES permits.

Milner Dam: Twin Falls, Shoshone Falls, Upper Salmon Falls, Lower Salmon Falls, and Bliss Dams (Figure 4-3). All five facilities are operated under licenses with the Federal Energy Regulatory Commission. In the Middle Snake River, there has been a 37% loss of free-flowing habitat (Cochnauer, 1983), a direct result of operating dams for hydroelectric power, flood control, and agricultural purposes. The fluctuations of water levels in impoundments, reservoirs, and tailwaters are both seasonal and diurnal in nature. Of these, the greatest change in water level in the Middle Snake River occurs during diurnal fluctuations in the tailwaters of a dam (Irving and Cuplin, 1956).

The change from a riverine system to a reservoir system is driven by the time water remains in one location. The longer the retention time, the more likely the system will function like a lake rather than a swiftly flowing stream. Retention times of the five reservoirs, at low river flow and average annual flow, are given in Table 4-4.

Low river flow for Twin Falls and Shoshone Falls are 5.66 cms (200 cfs) and 78.25 cms (2,763 cfs), respectively. Corresponding low flows for Upper and Lower Salmon Falls are 156 cms (5,510 cfs) and 254 cms (8,978 cfs) for Bliss. For the Middle Snake River assessment, Twin Falls and Shoshone Falls reservoirs were treated as reservoirs with the potential for vertical stratification, on the basis of data collected by the Idaho Power Company (Myers and Pierce, 1996).

Construction of impoundments destroys the natural geomorphological structure of the channel and mobilizes sediments. Stream flow regulation at hydroelectric dams can alter the upstream and downstream sediment distribution and thermal regime. Water released from dams results in increased erosion of the riverbed and banks below dams, particularly in the littoral areas. These habitats are most often altered in ways that are not compatible with the survival of

Table 4-4. Retention times for the five reservoirs in the Middle Snake River for low and average annual river flows

Hydroelectric project	Retention time at low river flow (days)	Retention time at average annual river flow (days)
Twin Falls	2.56	0.18
Shoshone Falls	4.57	0.33
Upper Salmon Falls	0.26	0.22
Lower Salmon Falls	0.83	0.51
Bliss	0.30	0.18

diverse native benthic communities. Downstream of the dams, the higher velocity discharges erode banks and the river bottom and carry suspended sediment to the backwaters of the next impoundment. The net result is deposition of suspended material upstream of a dam and scouring of the river bottom and shoreline areas downstream of the dam. Sediment transport capacities are lower upstream of impoundments because the velocity and turbulence of river currents is dissipated in the slowly moving backwaters of impoundments. Sediment scouring and deposition eliminate niches for species that prefer boulders or gravel and clear water (some invertebrates and fish species) and create niches for species that require sediment substrate for growth (rooted macrophytes).

The backwater upstream of these dams is slowed, warmed, and often stratified under relatively stagnant flow conditions. Falls or rapids in these areas are drowned by the elevated water surface upstream of the dam, and aeration capacity of the falls is lost.

Dams fragment a river system, isolating resident fish in tailwater reaches between them. Fish may be stranded and die in tailwater reaches, or they may be unable to reproduce because of inadequate habitats. The much longer hydraulic residence times permit development of planktonic algae and accumulation of soft bottom sediments, two conditions normally not associated with swift-flowing streams. Increased suspended sediments may also smother species or alter their behavior.

The annual range of water temperatures tends to fluctuate more because of the presence of the impoundments. The increase in surface area exposes more water to solar radiation, which tends to raise summer surface water temperatures. The combination of slower velocities and higher temperatures that results from dam operation creates an optimal environment for the growth of plankton and macrophytes.

4.11.7. Other Nonpoint Sources

Stream bank erosion (exacerbated by cattle grazing) and urban runoff also contribute sediments and nutrients to the river. There is minimal information on these sources in the Middle Snake drainage.

4.12. ECOSYSTEM DYNAMICS

To implement the dynamic model of mass and energy, the Middle Snake River has been divided into two major ecosystem components. One describes the chemical, physical, and biological characteristics of the moving water column, and the other describes the benthic plant community attached to or associated with the river bottom.

4.12.1. Water Column Dynamics

River water quality is high during most of the year, but may decline significantly at low flows in mid- and late summer. It is these extreme low-flow conditions that set bounds for aquatic species. In the fall with increasing water flows, water quality generally improves as less water is removed from the river channel and overland and subsurface irrigation return flows increase, especially below Niagara Springs. Most in-stream water quality parameters (with the exception of NO_3^-) decline in concentration through the fall (Myers et al., 1995).

The water quality of the river is strongly influenced by the natural springs. Alcove springs (springs discharging from the lower canyon walls along the Snake River banks) are common along the Middle Snake River below Twin Falls. These springs discharge exceptionally clear (Secchi disk transparency $\geq$ 10 m) and cool water that influences the water quality of the river channel.

However, these springs are not always removed from pollution. Clark and Ott (1996) estimated that the springs upstream of Twin Falls received more than 90% of their flow from irrigation recharge whereas the downstream springs receive less than 20% of their flow from irrigation recharge. As a result, conductivity and NO_3^- increase down through Rkm 957.4 (RM 595) and decrease from dilution below that point.

Nitrogen contributions to the Middle Snake River include nitrates in spring flows and limited instances of nitrogen fixation by blue-green algae. The State of Idaho completed a survey of groundwater of the Middle Snake River in 1991 (IDWR, 1992) and found that approximately 95% of the 129 sites monitored exhibited elevated levels of nitrate nitrogen. The springs' constant water temperatures, along with high conductivity and NO_3-abundant shallow depths, high alkalinity, high transparency, and hard-water conditions (Clark, 1997a), are all conducive to sustained high plant and invertebrate productivity in the springs proper; these inflows significantly influence water quality of the main river channel.

The range of nitrate nitrogen in the mainstem (Table 4-5) decreases in a downstream direction. At Rkm 985.7 (RM 612.6), the range is from less than 0.5 mg/L to more than 2.5 mg/L, while at Rkm 784.4 (RM 487.5) it is from about 1 to 2 mg/L. As in the case of temperature, this change can be attributed to the moderating effect of the springs on both flow and concentration. Natural levels of nitrogen were reported (Allen, 1995) as 0.12 mg/L dissolved inorganic nitrogen. Average phosphorus levels in the mainstem ranged from 0.06 to 0.17 mg/L for total phosphorus and 0.02 to 0.1 mg/L for ortho phosphate (Table 4-5).

In conjunction with the sampling of the invertebrates, several water chemistry variables were assessed at monthly intervals during the summer and autumn (Table 4-6).

Table 4-5. Average concentrations of nitrogen and phosphorus in the Middle Snake River (from Brockway and Robison, 1992)

Rkm (RM)	NH_4-N mg/L	NO_2+NO_3-N mg/L	Total P mg/L	PO_4-P mg/L
995 (619)	0.07	1.76	0.09	0.08
993.2 (617.3)	0.08	1.70	0.08	0.07
988.9 (614.6)	0.07	1.73	0.08	0.06
982.3 (610.5)	0.07	1.67	0.09	0.06
977.6 (607.6)	0.28	1.79	0.17	0.14
956.7 (594.6)	0.11	2.01	0.13	0.12
938 (583.0)	0.11	1.46	0.09	0.08
932.6 (579.6)	0.10	1.46	0.09	0.03
922 (573.0)	0.10	1.38	0.08	0.08
902.8 (559.9)	0.09	1.36	0.08	0.07

Table 4-6. Mean values for selected water chemistry variables from 1992 to 1994 in the Middle Snake River (see Royer et al., 1995, for full description)

Year	Alkalinity, mg $CaCO_3$/L	Hardness, mg $CaCO_3$/L	NO_2+NO_3 ppm	Total P, ppm	Specific conductance, S/cm
1992	195	243	1.90	0.119	535
1993	186	212	1.28	0.172	490
1994	188	218	1.54	0.151	449

Natural phosphorus levels in streams throughout the United States have been reported at 0.01 mg/L PO_4 (Allen, 1995). Concentrations of phosphorus exceeding 0.03 mg/L are generally indicative of eutrophication (Wetzel, 1983).

The ratio of nitrogen to phosphorus (Redfield, 1958) is 16:1 in plant tissue. In systems where the ratio falls below 16 it is assumed that nitrogen may be limiting. In the Middle Snake, the ratio ranges from >19 to 5 at Rkm 965.4 (RM 600) (Figure 4-9). Thus, the system at times may be both nitrogen and phosphorus limited. However, for most fresh waters phosphorus is assumed to be the driver for plant growth (Allen, 1995).

The abundant growth of aquatic macrophytes and filamentous algae, together with the high mean concentrations of nitrogen and phosphorus, indicate that the Middle Snake River was a highly eutrophic system during the years 1992-94. The eutrophic condition also was reflected in the extremely fast rate at which organic material in the river decomposed (Royer and Minshall, 1996).

Higher and later spring flows in 1993-94 were the cause of cooler temperatures those years. Optimal growth temperatures in the model were set at 25°C for both rooted and nonrooted forms. It is possible that nonrooted optimal growth temperature should be set lower to project greater growth at cooler temperatures. In the Box Canyon reach, where water temperatures were much cooler because of unpolluted springs influence, the modeled nonrooted growth was also much less than the observed growth.

An important characteristic of the water temperature in the Middle Snake River, as reflected in both the simulated and observed values, is the change in temperature range from upstream to downstream. At the location farthest upstream (Rkm 984.5, RM 612), water temperature varies from near 0°C to approximately 22°C. At the downstream locations the water temperature ranges from approximately 7°C to 20°C. The difference in water temperature range is due to the moderating effects of the spring flow on river temperatures in the lower reaches. Water temperature varies seasonally in the river, depending on meteorological conditions and groundwater flow. Average daily water temperatures at Rkm 893 (RM 555) vary seasonally from 5°C in winter to a maximum of approximately 21°C in summer (Figure 4-10). Average

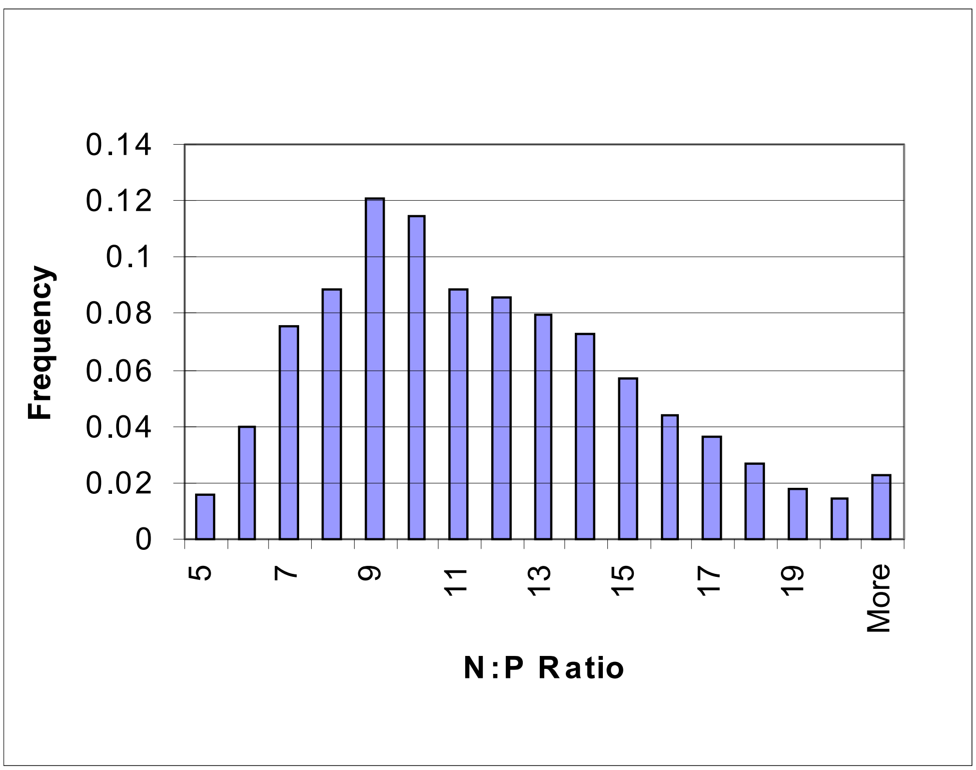

Figure 4-9. Frequency of N:P ratios in the Snake River at Rkm 965.4 (RM 600).

temperature for the springs was about 15°C; for the tributaries 12°C to 14.5°C. In 1994, monitoring of water temperature in the Middle Snake River near the Magic Valley Fish Hatchery (approximately Rkm 944, RM 590) revealed 40 days during July and August in which the mean daily water temperature exceeded 20°C.

The mathematical description of mass and energy flow in the water column described in this report is based on the mathematical model RBM10 (Yearsley, 1991). RBM10 has been used as a decision support tool in a number of river basins in the Pacific Northwest, including the Snake River above Milner Dam (Yearsley, 1976) and the Spokane River (Yearsley and Duncan, 1988). RBM10 makes use of concepts that have been used in other modeling efforts (e.g., Thomann et al., 1975; Patten et al., 1975; DiToro et al., 1975; Chen and Orlob, 1975; Scavia, 1980) and is conceptually similar to these models. Variables in the water column simulated by this model are given in Table 4-7.

The sediments with which the benthic plants are associated in the Middle Snake River are segmented into well-mixed compartments organized longitudinally only. In general (e.g., Ambrose et al., 1993), many of the physical, chemical, and biological processes in the sediments are similar conceptually to those in the water column. However, in this application of simulation methods to risk analysis, the analysis includes sediments only to the extent they provide substrate for benthic plants including vascular macrophytes, epiphytes, and periphyton. Sedimentation rates for phosphorus reported by Falter and Burris (1996) proved to be important to simulating changes in concentrations in the vicinity of maximum macrophyte density.

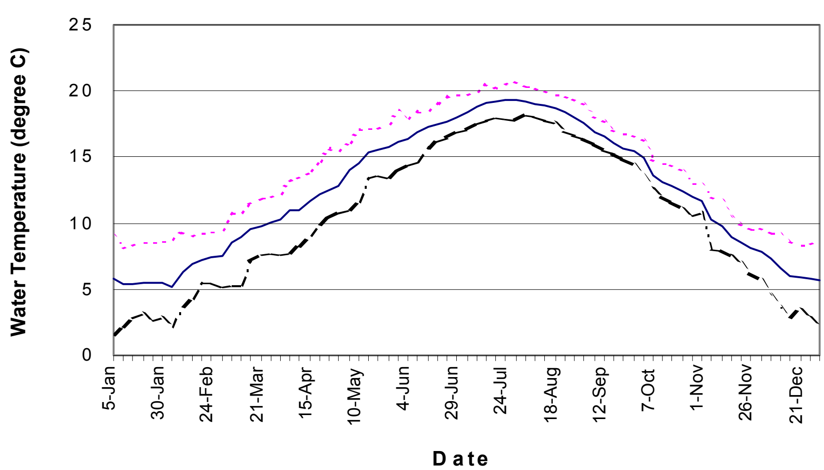

Figure 4-10. Simulated water temperatures (C) in the Middle Snake River at Rkm 893 (RM555); maximum, mean, and minimum.

The flow of mass and energy within the sediments is generally not included in this analysis. However, the flow of mass and energy within the water column as it affects uptake by the roots of vascular macrophytes is included in the analysis. Where flow of mass or energy from the sediments are part of the analysis, as in the case of nutrient flow to the roots of vascular macrophytes, it is assumed to be unlimited by plant uptake. Similarly, the flow of solids to and from the sediments is assumed to be at steady state. That is, there is neither gain nor loss of substrate due to deposition and scouring.

Chemical oxygen demand from point and nonpoint discharges does not appear to be a major source of oxygen demand in the Middle Snake River. Limited testing of the model showed that the dissolved oxygen in the river was not sensitive to changes in this parameter within the ranges typical of this system (Bowie et al., 1985).

4.12.2. Sediment Dynamics

Sediment dynamics in rivers and streams are driven by system hydrology and properties of sediment sources. In natural streams and rivers, a broad spectrum of both flows and sediment sources (Hill et al., 1991) shapes the character of the material that is transported in the river as suspended load or bed load, as well as shaping the character of the river channel form and bottom. When flow and sediment sources are the result of a broad spectrum of natural processes, river channels are characterized by a diverse ensemble of sediment types. Higher gradient river segments are typically ones with gravel, cobble, or boulder sediments. Smaller particle sizes ranging from sands to silts are associated with low-gradient segments or deep holes. The character of the substrate for the lower gradient segments is generally more transient as a result

Table 4-7. Water column variables simulated by the mathematical model for characterizing ecological risk

Carbonaceous biological oxygen demand (CBOD)	Organic nitrogen	Organic phosphorus	Coliform bacteria
Dissolved oxygen	Ammonia nitrogen	Orthophosphorus	Water depth
Phytoplankton biomass	Nitrite + nitrate nitrogen	Temperature	Water velocity

of high-flow events. Higher flow events are more likely to occur in a natural hydrologic regime, and such events are more likely to alter the river sediments in segments with smaller particle sizes.

In highly regulated systems, such as the Snake River, the spectrum of flows is modified considerably. As a result, the likelihood of high flows or rapid changes in flows is much less than that of the natural river. The construction and operation of hydroelectric facilities and the diversion of water for irrigation impose additional constraints on river hydraulics (Richter 1996). These constraints in turn can result in significant changes to both the channel geomorphology and substrate composition.

In the Middle Snake River, significant changes in the sources of sediments have also had a major impact on sediment dynamics. Field studies by Brockway and Robison (1992) found that the cumulative input of solids to the study reach from upstream sources during the period June 1990 to July 1991 near Milner Dam was approximately 3,400 tons, while the cumulative output from the study reach at King Hill was approximately 70,000 tons. During this same period, the input of solids to the segment between Rock Creek (Rkm 971.2) and the Gridley Bridge (Rkm 938.0) was 14,800 tons from the Snake River, 14,400 tons from irrigation canals, 44,500 tons from tributaries draining irrigated agriculture, and 4,100 tons from the major aquaculture facilities. Approximately 48,700 tons of solids was output from the study reach at the Gridley Bridge, leaving an excess of approximately 29,100 tons of fine-grained solids that were presumably deposited in the study reach during this period.

High deposition rates of fine-grained solids in the Middle Snake River during this period were confirmed in the results of field studies reported by Platts (1991). Platts (1991) separated substrate types in the reach from Rkm 967.0 to Rkm 951.7 into nine categories: silt, silt and sand, sand, sand and gravel, gravel, cobble, boulder, boulder and bedrock, and bedrock. Platts (1991) found that silts made up 56.7% of the area surveyed. Studies of sediment chemistry by Falter and Burns (1996) in this same segment found that organic matter in the surficial sediments varied between 2.6% and 4% and that average phosphorus concentrations varied between 1,073 and 1,577 mg/g. These sediments provided an ideal substrate for the luxuriant growths of macrophytes and epiphytes observed in this segment of the Middle Snake River.

The period during which high deposition rates of fine-grained, nutrient-rich sediments were observed in the study reach was a period of extremely low flows. In 1997, flows in the study reach were extremely high. Two studies of sediments and sediment transport conducted during the summer of 1997 (Clark, 1997a; McLaren, 1998) provided insight into how channel morphology and deposition rates in the regulated river respond to high flow conditions. McLaren (1998) found that deposition during this period was occurring only in Shoshone Falls Reservoir, the most upstream reservoir included in this study. Downstream from Shoshone Falls, percentage of fine particles increased in a downstream direction. McLaren (1998) concluded that most of the suspended sediments transported by the river were derived principally from the river bed itself and that the increase in fine-grained sediments in the downstream direction was a result of the natural progression of size sorting in the direction of sediment transport. Because of the manner in which sediments had been scoured from the study reach, McLaren (1998) likened the river to a “chute contained in bedrock.” These conclusions were supported by the work of Clark (1997), who found that in some areas of the river, the bed sediment material was hard-packed sand, essentially impervious to penetration. Clark (1997) also found that the only segment of the system that had fine-grained clay-sized sediments was near Bliss Dam at Rkm 901.5, supporting McLaren’s (1998) hypothesis that fine-grained sediments had moved downstream.

The picture of sediment dynamics in the Middle Snake River that emerges from these studies is one in which the ends of the spectrum of sediment processes are represented, but there is no continuum. That is, the processes that might lead to a more natural system have been impaired by changes in the river hydrology and sediment inputs. This results in a system with predominantly high-organic, nutrient-rich, fine sediments during periods of low flows. Most of these sediments come from land-use practices related to irrigated agriculture and aquaculture. When river flows are high enough to scour the fine-grained particles, which McLaren (1998) suggests is at about 283 cms (10,000 cfs), the sediments are transported downstream. Some of these sediments are deposited in the downstream reservoirs, as evidenced by the organic sediments found by Clark (1997) at Bliss Dam. In river segments outside of the reservoirs, a scoured channel composed of hard-packed sands and bedrock may occur because there is a lack of connection with upstream, natural sources of sediment. This condition can be made worse during periods of high river flows.

4.12.3. Dynamics of the Benthic Plant Community

The benthic plant community variables included in the analysis for the sediments are macrophytes with roots, macrophytes with limited roots, epiphytes, and periphyton. The flow of energy, mass, and information for the benthic plant community is shown in Figure 4-11.
The concept for kinetics of vascular macrophytes is based on the terrestrial ecosystem energy model developed by O'Neill et al. (1972) for a closed-canopy, homogeneous forest ecosystem in the eastern deciduous biome. Bloomfield et al. (1973) adapted the concept to simulate aquatic macrophytes in Lake George, New York. The analysis for aquatic macrophytes in the Middle Snake River is similar to the Lake George model. Important features of this concept are:

- The organic matter associated with vascular macrophytes can be idealized by three compartments for organic carbon, including roots, leaves/shoots, and carbon storage.

- The accumulation of carbon in carbon storage is by photosynthesis. Carbon flows to roots and leaves/shoots from storage.

Other features of the analysis for vascular macrophytes are based on previous research and observations of macrophytes in the Middle Snake River (Falter and Carlson, 1994; Falter et al., 1995; Falter and Burris, 1996). These features are characterized by several assumptions:

- Michaelis-Menten formulations are appropriate for light, nutrient, and habitat limitations (e.g., Barber, 1991; Porcella et al., 1983).
- Nutrient uptake rates are low at low river velocities because of poor rates of exchange, but increase with river velocity up to a certain optimal velocity (Horner et al., 1983). As river velocity increases beyond a certain point, physical stresses begin to occur in the plants. These stresses lead to mortality of the plants and increase the rate of sloughing (Chambers et al., 1991a,b).
- Vascular macrophytes with extensive root systems, such as *Potamogeton,* take a

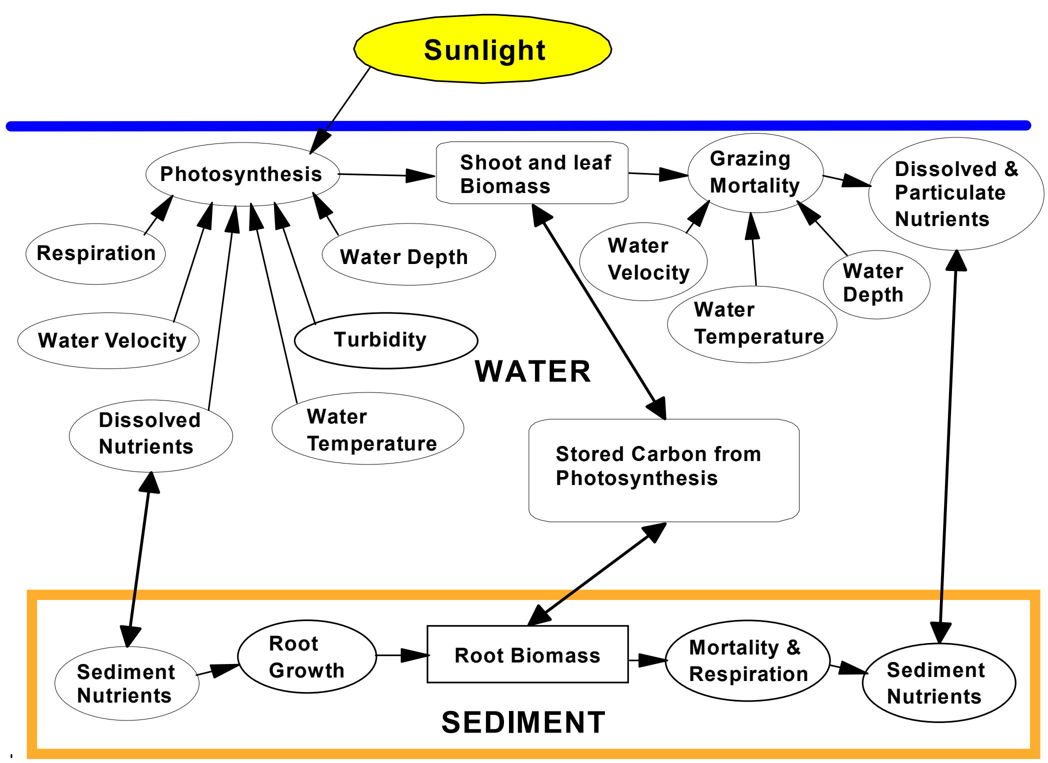

Figure 4-11. Flow of energy and materials for aquatic plant growth in the Middle Snake River.

large percentage of their nutrients from the sediments (Howard-Williams and Allanson, 1981). Macrophytes with limited root systems, such as *Ceratophyllum*, derive the majority of their nutrients from the water column.

The analysis for epiphytes is a population model with Michaelis-Menten formulations for light, nutrient, and habitat limitations. This analysis was modified to include the assumption that, in the Middle Snake River, epiphytes such as *Cladophora* are generally associated with a macrophyte substrate on which they attach themselves and grow. Furthermore, the epiphytes intercept solar radiation in the top 10% of the water column, rather than over the entire water column. This assumption was based on observations made during the 1992-1994 studies of macrophytes (Falter and Carlson, 1994; Falter et al., 1995; Falter and Burris, 1996).

Rates of phytoplankton growth, respiration, nutrient, light, and temperature limitations and stoichiometry were initially based on values typical of those used in other phytoplankton model studies (Bowie et al., 1985). Sensitivity analysis showed the dynamics of phytoplankton in the Middle Snake River to be more responsive to hydrology and to initial conditions from upstream sources.

Rooted aquatic macrophytes in the Middle Snake appear to be most responsive to water depth and sediment composition. These two factors are controlled by sediment loading (primarily from agricultural drains and fish hatcheries) and localized hydrology. Once physical structure is provided, nonrooted macrophtyes can develop, given sufficient structure and nutrient supply from the water column.

In addition to the parameters characterizing mass and energy transfer, a benthic habitat factor was introduced. The benthic habitat factor was an estimate of the fraction of the bottom area available for macrophyte growth in each river segment. Downstream of Auger Falls this factor was estimated from the macrophyte studies conducted by Hill (1992). Above Auger Falls the habitat factor for macrophytes was assumed to be zero, primarily because of lack of data.

Initial estimates for the parameters characterizing growth rates, rates of senescence, and nutrient uptake were varied by trial and error using mass and energy loading as described by 1990-1994 water chemistry and hydrology data given above and 1992-1994 macrophyte data reported by Falter and Carlson (1994).

5. SIMULATION OF ECOLOGICAL RISK

The following chapters (5, 6, 7, and 8) present the results of the analysis of exposure and effects on the assessment endpoints described in Chapters 3 and 4. The analysis begins with the results of the simulation of ecological events (Chapter 5) in the basin. This is followed by a qualitative analysis of the factors affecting fish (Chapter 6), invertebrates (Chapter 7), and plants (Chapter 8).

The methods and results for the simulation of ecological risk are provided in Appendix D of this report. A summary of the quantitative measures of effect and risk estimates is presented in this chapter.

5.1. QUANTITATIVE MEASURES OF EFFECT

5.1.1. Water Quality Standards

A logical source for measures of effect is the State of Idaho's water quality standards (Table 5-1). The standards for dissolved oxygen, temperature, and ammonia are numerical thresholds based on reviews of the literature. The standards for nutrients and macrophytes are narrative. In order to estimate risks quantitatively, these narrative standards must be converted to numeric limits.

The State of Idaho's Department of Environmental Quality (IDEQ) derived a numeric criterion for phosphorus for the purposes of establishing a total maximum daily load (TMDL). The TMDL requires that total phosphorus be 0.075 mg/L when the river flow is equal to the 1-in-10-year 7-day average low flow (7Q10), as measured at the Gridley Bridge. The TMDL limits total phosphorus in the Middle Snake River at the Gridley Bridge to 1,088.6 kg/day. This corresponds to a concentration of 0.075 mg/L total phosphorus for a flow of approximately 167 m^3/s (5,900 cfs) in the Snake River. This criterion is less than that suggested by U.S. EPA (1976) for flowing waters (0.10 mg/L), but greater than that suggested by U.S. EPA (1976) for flowing waters that enter lakes or reservoirs (0.05 mg/L). The criterion for total phosphorus developed by IDEQ has not been specifically related to the levels of macrophyte growth in the river that would exceed the State of Idaho's narrative water quality standard for nutrients.

In an effort to define a quantitative measure for nuisance levels of aquatic macrophytes, a literature survey was conducted (Appendix D). Only those papers that made reference to water quality impacts and had quantitative data for macrophyte biomass were used to develop the measurement endpoint. Types of water quality impacts included general water quality

Table 5-1. Variables simulated by the dynamic model and their associated measures of effect (State of Idaho water quality standards and habitat suitability factors) and assessment endpoints

Variable	Measures of effect	Assessment endpoint
Dissolved oxygen	State of Idaho Water Quality Standards: 6 mg/L except in: (1) bottom 20% of lakes or reservoirs with water depths < 35 m, (2) bottom 7 meters with water depths > 35 m, and (3) hypolimnion of stratified lakes or reservoirs	Reproduction and survival of cold-water biota
Water temperature	State of Idaho Water Quality Standards: equal to or < 22^{O}C, maximum daily average equal to or < 19ºC. Habitat suitability factors	Reproduction and survival of cold-water biota
Dissolved oxygen and water temperature	1-day minimum DO is not < 90% saturation. Water temperatures equal to or < 13°C and maximum daily average < 9ºC	Salmonid spawning and incubation periods: Rainbow trout: Jan 15 to July 15 Mountain whitefish: Oct 15 to Mar 15
Total phosphorus	State of Idaho phosphorus TMDL target of 0.075 mg/L	Growth of vascular macrophytes and algae
Water depth	Habitat suitability factors	Reproduction and survival of cold-water biota
Water velocity	Habitat suitability factors	Reproduction and survival of cold-water biota
Nutrients	Macrophyte biomass < 200 g/m^2	Surface water of the State shall be free from excess nutrients that can cause visible slime growth or other nuisance aquatic growths impairing designated beneficial uses
Un-ionized ammonia	State of Idaho Water Quality Standards	Reproduction and survival of cold-water biota

degradation, aquatic environment alteration, and eutrophication. The results of this survey (Table 5-2) suggest that an average maximum biomass of 200 g/m^2 as ash-free dry matter (AFDM) is a reasonable lower bound for nuisance levels of aquatic macrophytes.

For the Middle Snake River, there are site-specific measures of effect that can be integrated into the analysis plan to provide additional lines of evidence. Among these are indices the U.S. Fish and Wildlife Service (USFWS) developed to characterize habitat suitability for certain cold-water species.

Table 5-2. Maximum biomass of macrophytes in water bodies with water quality problems

Species	Biomass	Impact	Reference
Heterantherea dubia *Myriophyllum spicatum* *Potamogeton* sp.	150 to 275 g/m^2	Water quality degradation	Barber, 1991
Myriophyllum spicatum *Potamogeton crispus* *Elodea canadensis*	Standing crop 350 to 900 g/m^2	Eutrophic lake	Nichols and Shaw, 1986
Hydrilla verticillata	200 to 800 g/m^2 ODW[a]	Altered aquatic environment	Bowes et al., 1979
Ceratophyllum demersum	~ 200 to 800 g/m^2	Eutrophic lake	Westlake, 1963
Ceratophyllum demersum *Potamogeton pectinatus*	300 g/m^2 ODW	Eutrophic lake	Filbin and Barko, 1985
Ceratophyllum demersum *Myriophyllum spicatum* *Potamogeton* sp. *Chara vularis* *Cladaphora fracta* *Utricularia vulgaris*	250 g/m^2 AFDM[b]	Eutrophic lake	Hough et al., 1989
Myriophyllum spicatum *Ceratophyllum demersum* *Potamogeton pectinatus*	300 to 600 g/m^2 AFDM	Mesotrophic slow flowing	Falter et al., 1991
Lyngbya birgei	120 to 1,300 g/m^2 ODW	Major nuisance growth	Beer et al., 1986
Scapania undulata *Marsupella aquatica* *Fontinalis dalicarlica* *F. antipyretica* *Bulbochaetae sp.* *Microspora sp.* *Mougeotia sp.* *Zygnema sp.*	140 to 670 g/m^2 Dry weight	Regulated stream with reduced river amenity	Rorslett and Johansen, 1995

[a]ODW is oven dry weight.
[b]AFDM is ash-free dry matter.

5.1.2. Habitat Suitability Indices

Although the water quality standards do provide measures of exposure and effect, they typically relate to a fairly broad range of aquatic species and environments. For example, the water quality standards define certain criteria for the protection of cold-water species without specifically defining the set of cold-water species. These criteria may be protective of aquatic organisms within the group characterized as cold-water species, but there well may be some organisms or certain life stages of organisms in this group that are more vulnerable. Because of this, it is desirable to develop site-specific lines of evidence if they are available. The USFWS uses habitat suitability indices to assess the impacts of flow modification on aquatic habitat resources of rivers and streams. An important objective of this method, called the Instream Flow Incremental Method (IFIM), is to make quantitative comparisons of habitat conditions at differing regimes of river flow.

The USFWS, in support of the IFIM, has developed habitat curves for many aquatic organisms. In a cooperative study conducted with the Idaho Power Company, Anglin et al. (1992) describe habitat suitability curves for cold-water fishes. The periods of the year to which these suitability indices apply are shown in Table 5-3.

In applications of IFIM, the habitat suitability curves are used to develop flow-weighted measures of habitat suitability. In this ecological risk assessment, the simulation results from the ecological model were compared with the IFIM habitat suitability curves. The frequency with which the simulated results were less than a reasonable value of the habitat suitability curve was used to assess whether the system would support a particular life stage of the target organism. The reasonable level for habitat suitability was defined as 0.6. That is, values of the habitat index greater than 0.6 were assumed to be representative of conditions supporting the particular life stage of the target organism, whereas values of the index less than 0.6 were assumed to be representative of conditions that would not support that life stage. The criterion of 0.6 was chosen simply because it is slightly greater than 0.5. Although this was somewhat arbitrary, the estimates of ecological risks are not particularly sensitive to the criterion, given the shapes of the habitat suitability curves. Most of the uncertainty in the estimates of ecological risks using these habitat suitability curves is in the shapes of the curves.

Quantitative comparisons are accomplished in the IFIM by calculating habitat suitability for various regimes of river flow. Integral components of this calculation are the habitat suitability curves. These curves define suitability indices for different life stages of target aquatic species selected for a particular study. The suitability indices are functions of ecosystem variables such as water depth, water velocity, water temperature, and substrate or cover type.

Because the indices measure suitability of habitat as a function of the condition of the ecosystem, they can also be used as measures of effect. However, the target species for which suitability is quantified must be relevant in terms of the assessment endpoints. In the case of the

Table 5-3. Time of the year for which habitat factors are applied to various life stages of cold-water fish native to the Snake River (from Anglin et al., 1992)

	Jan Feb Mar Apr May Jun Jul Aug Sept Oct Nov Dec
Rainbow trout	
Spawning	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
Incubation	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
Fry	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
Juvenile & adult	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
Mountain whitefish	
Spawning	xxxxxxxxxxxxxxxxxxxxxxxxxxxx
Incubation	xxxxxxxxxxxxxxxxxxxxxxxxxxxxx xxxxxxxxxxxxxxxxxxxxxxxxxxxx
Fry	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
Juvenile & adult	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
White sturgeon	
Spawning	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
Incubation	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
Larvae	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
Passage	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
Juvenile & adult	xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx

Middle Snake River, this means they must be cold-water species that are native to the region. On the basis of the work of Anglin et al. (1992), three cold-water fishes found in the Middle Snake River, the mountain whitefish, rainbow trout, and white sturgeon, have been selected as target species to be used with the habitat suitability indices. Anglin et al. (1992) developed habitat suitability indices for these species for the Snake River from C.J. Strike Dam downstream to the upper end of Brownlee Pool. Because habitat suitability indices appropriate for large river systems were lacking for these species, Anglin et al. (1992) used criteria from smaller river systems. Extension of the criteria from smaller river systems to the Snake River was done subjectively and based on the judgment of regional biologists. For the purposes of the ecological risk analysis, we have assumed the process adequately represents the habitat requirements for the target species in the Middle Snake River.

For cold-water biota, the ecological variables, measures of effect, and assessment endpoints used in this simulation are shown in Table 5-1.

5.2. RISK ESTIMATES

5.2.1. Exceedance of Water Quality Standards

Ecological integrity of an aquatic ecosystem depends on the characteristics of the water temperature and dissolved oxygen regimes. To characterize stress associated with temperature and dissolved oxygen, the simulated 67-year record of variables is compared with the water quality criteria in each of the representative segments. The comparison is made for the general category of cold-water species and for spawning of two species, the mountain whitefish and the rainbow trout.

Stress occurs when the temperature-dissolved oxygen envelope experienced by the target organism is larger than the envelope associated with its physiological requirements. Superimposing the envelope for water temperature and dissolved oxygen given in the water quality standards for each of these groups on the simulated temperature-DO diagram is a way of assessing stress associated with the temperature-DO regime in a particular segment. The frequencies with which the simulated values fail to fall within the envelope for temperature and DO defined by the State of Idaho's water quality standards are given in Table 5-4.

5.2.1.1. *Dissolved Oxygen*

For cold-water biota, the frequencies with which the simulated daily-averaged DO falls outside the envelope defined by the water quality standards is less than 0.01 in all of the

Table 5-4. Frequency with which simulated values of water temperature and dissolved oxygen (DO) are outside the envelope of the State of Idaho's water quality standards for cold-water biota, spawning rainbow trout and mountain whitefish

	Cold-water biota			Rainbow trout			Mountain whitefish		
Study reach segment	**Temp., daily avg.**	**Temp., max.**	**DO, daily avg.**	**Temp., daily avg.**	**Temp., max.**	**DO, daily avg.**	**Temp., daily avg.**	**Temp., max.**	**DO, daily avg.**
Milner Dam to Twin Falls	0.19	0.13	0.01	0.58	0.48	0.57	0.05	0.02	0.06
Twin Falls to Shoshone Falls	0.20	0.11	0.01	0.60	0.47	0.58	0.06	0.01	0.05
Shoshone Falls to RM 609	0.17	0.02	0.01	0.63	0.49	0.56	0.07	0.02	0.03
RM 609 to Rock Creek	0.15	0.01	0.01	0.64	0.50	0.55	0.08	0.03	0.03
Rock Creek to Crystal Springs	0.17	0.14	0.01	0.62	0.53	0.54	0.07	0.03	0.02
Crystal Springs to Boulder Rapids	0.13	0.01	0.01	0.66	0.52	0.37	0.10	0.03	0.01
Boulder Rapids to Kanaka Rapids	0.13	0.01	0.01	0.67	0.50	0.26	0.10	0.02	0.01
Kanaka Rapids to Gridley Bridge	0.04	0.01	0.01	0.74	0.53	0.54	0.17	0.03	0.11
Gridley Bridge to Upper Salmon Falls	0.01	0.01	0.01	0.80	0.54	0.19	0.22	0.04	0.01
Upper Salmon Falls to Lower Salmon Falls	0.03	0.01	0.01	0.78	0.52	0.26	0.20	0.03	0.01
Lower Salmon Falls to Bliss Bridge	0.05	0.01	0.01	0.69	0.48	0.42	0.11	0.01	0.01
Bliss Bridge to Bliss Dam	0.07	0.01	0.01	0.69	0.48	0.44	0.11	0.01	0.01
Bliss Dam to King Hill	0.11	0.01	0.01	0.68	0.48	0.33	0.10	0.01	0.01

segments. Frequencies for spawning mountain whitefish are less than 0.06 throughout the Middle Snake River, except in the segment from Kanaka Rapids to Gridley Bridge, where the frequency is 0.11. The frequencies for spawning rainbow trout range from 0.26 to 0.57 between Milner Dam and the Gridley Bridge, and 0.19 to 0.45 between the Gridley Bridge and King Hill. The higher frequencies associated with spawning rainbow trout are due to the fact that applicable water quality standards include some summer months, whereas the water quality standards for spawning mountain whitefish apply to fall and winter months when saturation levels of DO are higher.

5.2.1.2. *Temperature*

The frequency with which the daily-average simulated water temperature falls outside the envelope for cold-water biota ranges between 0.13 and 0.20 for the segments between Milner Dam and Kanaka Rapids. The frequency decreases to less than 0.05 from Kanaka Rapids to the Bliss Bridge because of the large volume of cooler water supplied by the spring flow. Between the Bliss Bridge and King Hill, the frequency increases slightly as the spring flow decreases and the transfer of thermal energy across the air-water interface becomes more important.

The pattern for the frequencies with which the simulated maximum water temperatures fall outside the envelope for cold-water biota is similar to that of the simulated daily-averaged water temperature. The decrease in frequency occurs in the Crystal Springs to Boulder Rapids segment, slightly upstream from the Boulder Rapids to Kanaka Rapids segment, in which the frequency of daily-averaged water temperatures decreases. In addition, the magnitude of the frequencies for which the maximum temperatures fall outside the envelope for cold-water biota is less than for the daily-averaged temperatures. The frequencies for maximum simulated water temperatures varied between 0.01 and 0.13 from Milner Dam to Boulder Rapids and were less than 0.01 between Boulder Rapids and King Hill.

The frequencies with which simulated daily-average water temperatures fall outside the envelope for spawning rainbow trout is greater than 0.58 throughout the Middle Snake River and greater than 0.48 for simulated maximum daily water temperatures. As in the case of DO, the high frequency of excursions above the criterion is due to the fact that the water quality standards for spawning rainbow trout include some summer months.

Frequencies of daily-averaged temperatures falling outside the envelope for spawning mountain whitefish are lowest in the upstream segments between Milner Dam and Kanaka Rapids and highest in the segments between Kanaka Rapids and Bliss Dam. Frequencies for daily-averaged simulated water temperatures vary between 0.05 and 0.10 in the upstream segments and between 0.10 and 0.22 in the downstream segments. The relatively low frequency of excursions above the criterion is due to the fact that water quality standards for spawning mountain whitefish apply to fall and winter periods when water temperatures are low.

5.2.1.3. *Phosphorus*

The simulation results from the dynamic model were used to obtain an empirical cumulative distribution function for total phosphorus at locations representing the segments given in Table 1. The empirical cumulative distribution function gives the probability that the simulated total phosphorus is equal to or less than some specified value. These functions represent stressor characteristics for existing levels of management.

In the segments of the Middle Snake River upstream from major point source and nonpoint source inputs, the estimated probability that total phosphorus will be equal to or less than 0.075 mg/L is between 0.23 and 0.25. In the segments between Rock Creek and Crystal Springs, total phosphorus loads from the City of Twin Falls STP, fish hatcheries, and irrigation return flows reduce the probability that total phosphorus will be equal to or less than 0.075 to between 0.01 and 0.04 (Figure 5-1). Between Bliss Dam and King Hill, the large volume of spring inflow with low levels of total phosphorus increases the estimated probability to 0.2-0.6 (Figure 5-2). All the cumulative frequency distributions are included in Appendix D.

5.2.1.4. *Macrophyte Biomass*

The cumulative distribution function for total macrophyte and epiphyte biomass was estimated for the Crystal Springs to Boulder Rapids segment. This segment was chosen because it has had among the highest levels of macrophyte growth measured during field studies by the University of Idaho (Falter et al., 1995; Falter and Burris, 1996). The probability that the simulated values of macrophyte biomass, measured as the sum of rooted macrophytes, nonrooted macrophytes, and epiphytes, would be less than 200 g C AFDM/m^2 was estimated to be less than 0.01 for the 67-year period of record (Figure 5-3).

Simulated levels of un-ionized ammonia were compared with the chronic and acute criteria of the State of Idaho's water quality standards. The frequency with which the simulated levels exceeded the criteria was computed as the ratio of the number of values that exceeded a criterion divided by the number of simulated values. The estimated frequency with which the simulated values of un-ionized ammonia were below the chronic and acute criteria was estimated to be less than 0.05 and 0.01, respectively, throughout the Middle Snake River.

5.2.2. Habitat Suitability Indices

Measures of stress to the target cold-water species—mountain whitefish, rainbow trout, and white sturgeon—attributable to water temperature and hydrologic effects were obtained from habitat suitability indices developed by the USFWS (Anglin et al., 1992). The measures of habitat suitability associated with water temperature, water depth, and velocity were obtained for each life stage of the target organism and determined in the following way.

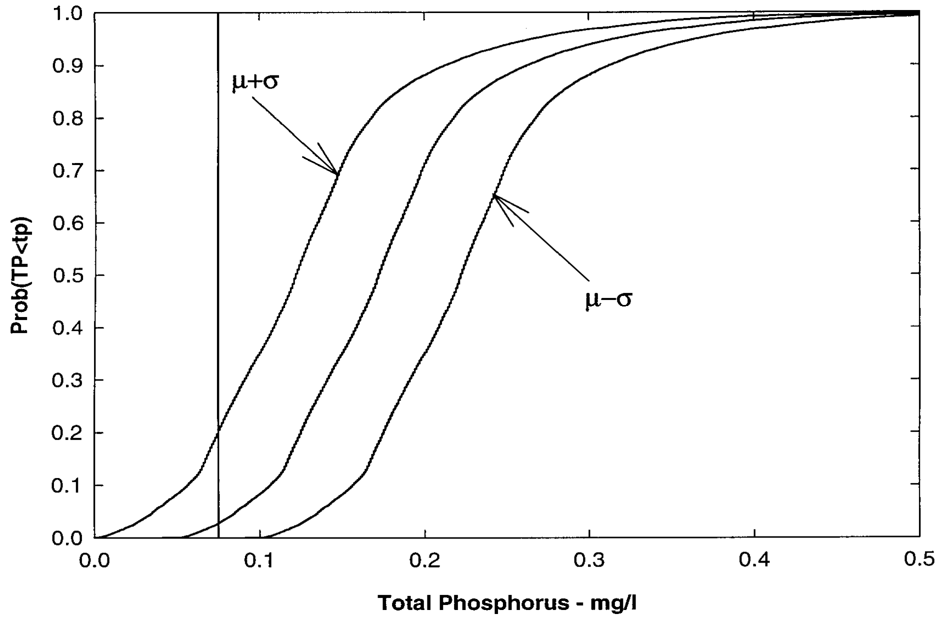

Figure 5-1. Cumulative distribution function for total phosphorus, Rock Creek to Crystal Springs. Results are for: (1) mean of simulated values, (2) mean minus one standard deviation, and (3) mean plus one standard deviation.

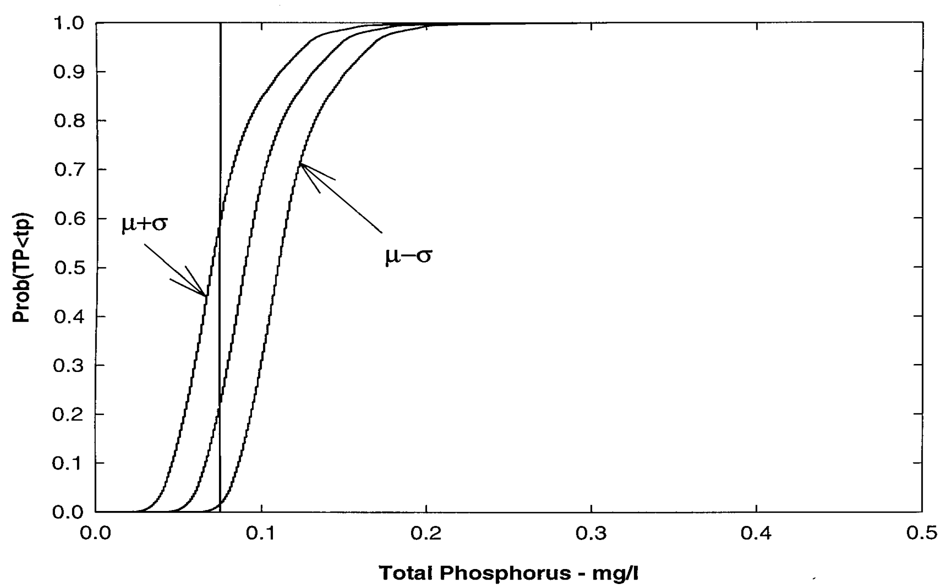

Figure 5-2. Cumulative distribution for total phosphorus, Bliss Dam to King Hill. Results are for: (1) mean of simulated values, (2) mean minus one standard deviation, and (3) mean plus one standard deviation.

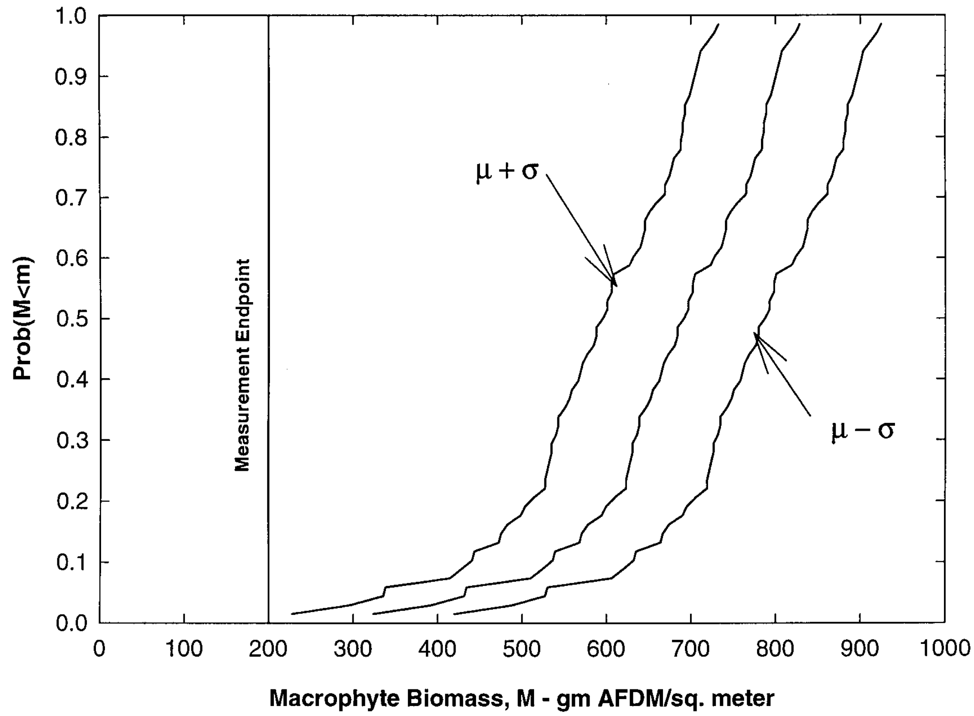

Figure 5-3. Cumulative distribution function for simulated macrophyte biomass in the Snake River at Rkm 965.4 (RM 600). Results are for (1) mean of simulated values, (2) mean minus one standard deviation, and (3) mean plus one standard deviation.

The index of impairment for each life stage was computed as the ratio of the number of simulated days in which impairment of habitat occurred (when the habitat index fell below 0.6) to the total number of days for which the life stage was vulnerable (Table 5-3). Four categories of impairment were defined based on the index of impairment:

1. less than or equal to 0.1, lowest impairment;
2. greater than 0.1 and less than or equal to 0.5;
3. greater than 0.5 and less than or equal to 0.9;
4. greater than 0.9 and less than or equal to 1, greatest impairment.

The index of impairment was generally high (Figure 5-4) throughout the Middle Snake River for all life stages of rainbow trout. Estimated levels of impairment for adults were estimated to be moderate to low in some portions of the segments from Rock Creek to Crystal

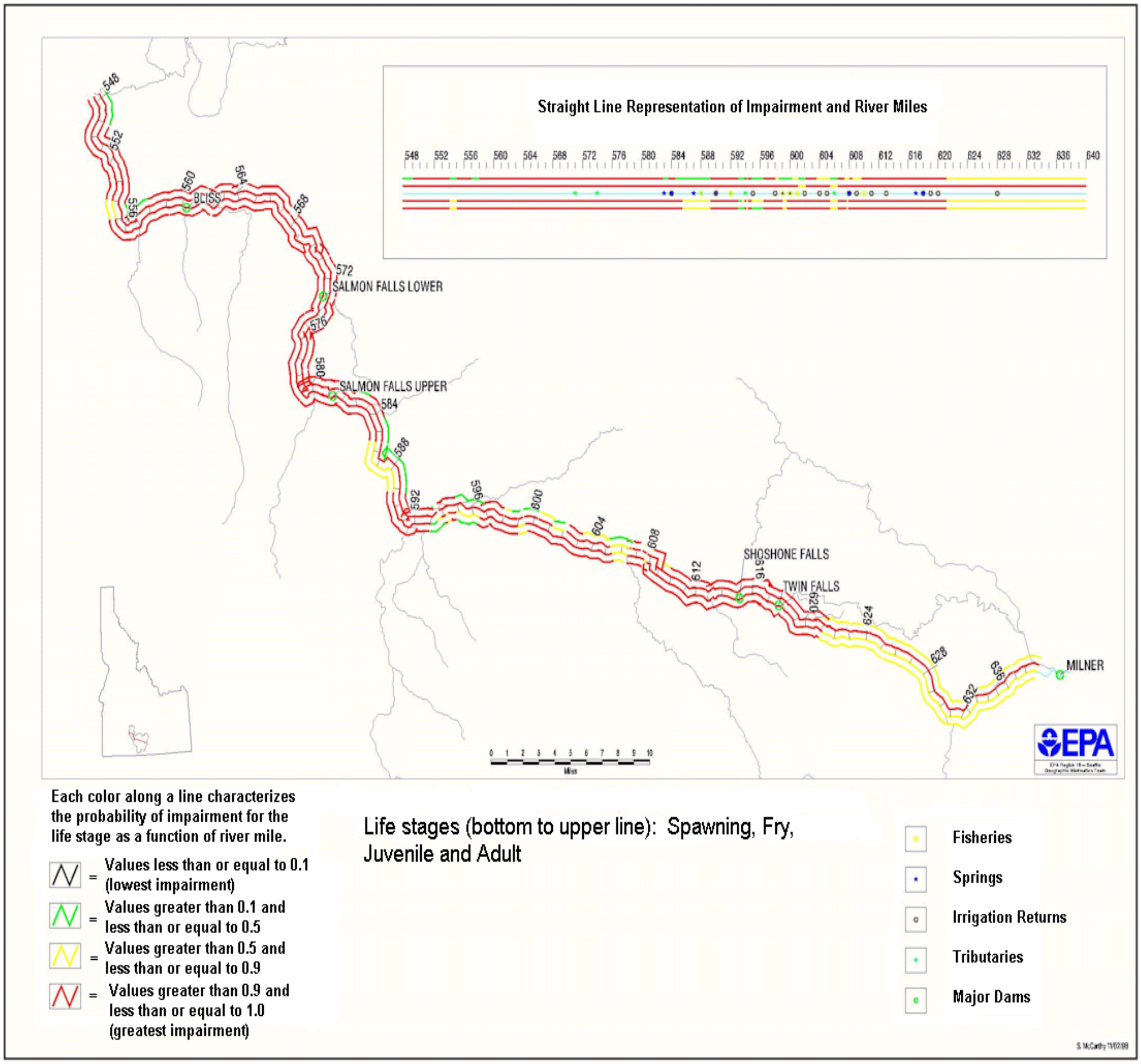

Figure 5-4. Probability of life stage impairment for rainbow trout in the Middle Snake River.

Springs, from Boulder Rapids to Kanaka Rapids, and in the upper portion of Upper Salmon Falls reservoir. Estimated levels of impairment for spawning were low to moderate in some portions of the segment from Boulder Rapids to Kanaka Rapids, as were estimated levels of impairment for rainbow trout fry.

Estimated values of the index of impairment for spawning, fry, and juvenile mountain whitefish were high throughout the Middle Snake River (Figure 5-5). For adult mountain whitefish, the values of the index of impairment were estimated to be moderate to high throughout the Middle Snake River. Most favorable conditions for adult mountain whitefish (moderate impairment) were found in Upper Salmon Falls and Lower Salmon Falls Reservoirs. These somewhat favorable conditions were primarily a result of high-quality, cooler water entering the Middle Snake River from springs.

Above Lower Salmon Falls Dam, estimated values of the index of impairment for white sturgeon were generally high for all life stages (Figure 5-6). Exceptions were found in the pool below Auger Falls and in Upper Salmon Falls Reservoir, where the index of impairment for adult white sturgeon was estimated to be low and the index of impairment for juveniles was to estimated to be moderate. Portions of the Wiley reach (Lower Salmon Falls Dam to the Bliss Bridge) had low to moderate values of the index of impairment for spawning and larval stages. The most favorable conditions for all life stages of white sturgeon were estimated to occur in the Middle Snake River below Bliss Dam.

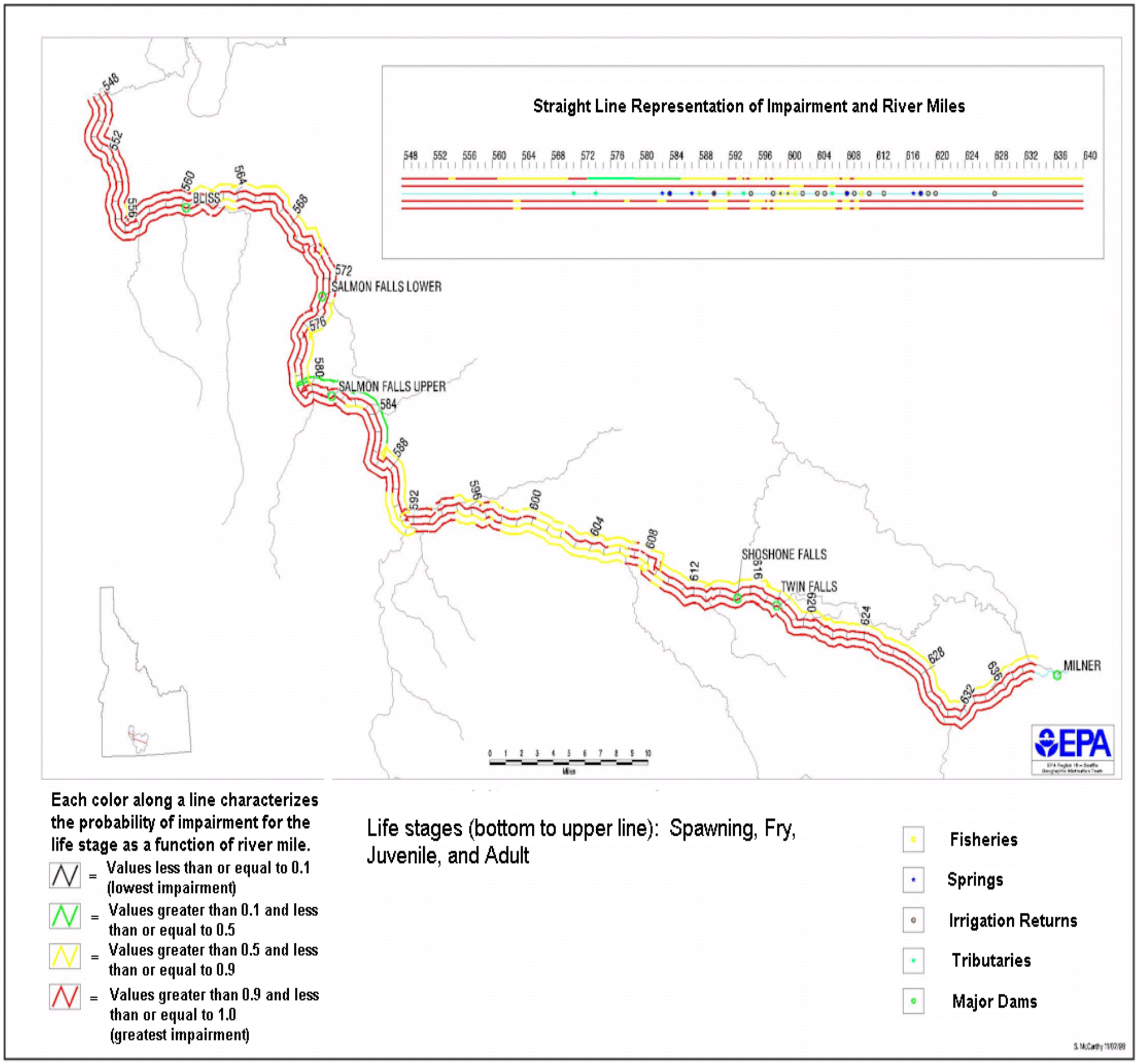

Figure 5-5. Probability of life stage impairment for mountain whitefish in the Middle Snake River.

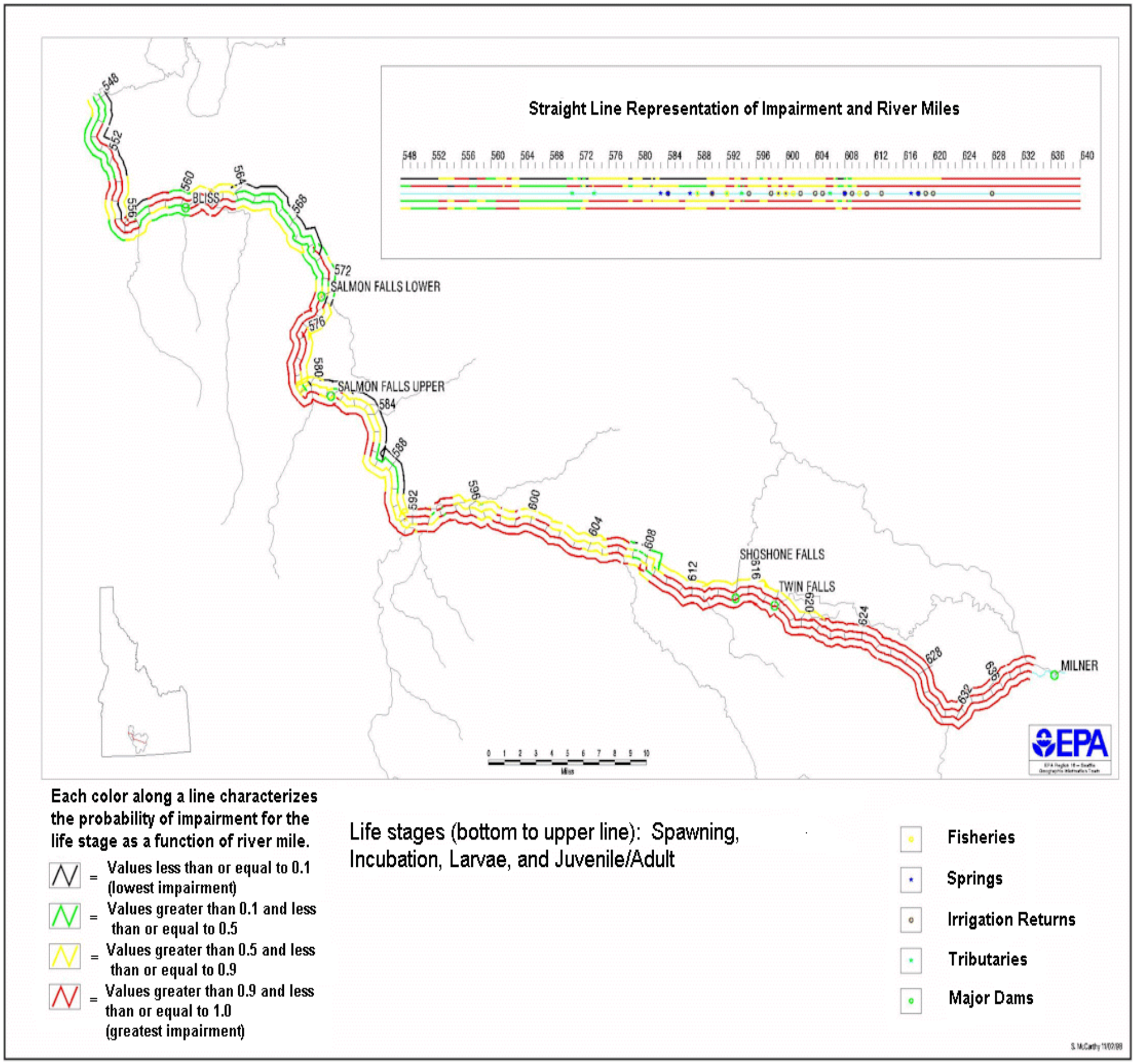

Figure 5-6. Probability of life stage impairment for white sturgeon in the Middle Snake River.

6. ANALYSIS OF EXPOSURE AND EFFECTS FOR THREE FISH POPULATIONS

The objective of this assessment is to provide a forecast for the long-term natural reproductive survival of rainbow trout (*Oncorhynchus mykiss*), mountain whitefish (*Prosopium williamsoni*), and white sturgeon (*Acipenser transmontanus*) in the Middle Snake River. The assessment will focus on rainbow trout and mountain whitefish from King Hill to Milner Dam and white sturgeon between C.J. Strike and Bliss Dams (the Bliss reach), between Bliss and Lower Salmon Falls Dams (the Lower Salmon Falls reach), and between Upper Salmon Falls Dam and Shoshone Falls (the Shoshone Falls reach). Loss and alteration of lotic habitats, increased water temperatures, and other stressors such as sedimentation that can directly and indirectly adversely affect these fish populations are discussed.

6.1. RAINBOW TROUT *(Oncorhynchus mykiss)*

The rainbow trout is the most important game fish in the Middle Snake River (Dey and Minshall, 1992). As the numbers of wild rainbow trout are at low levels (Cochnauer, 1980a, 1981; Lukens, 1982), hatchery fish have been stocked throughout this reach for many decades. Steelhead, the anadromous form of this species, originally inhabited this area, but dam construction for irrigation and power generation has reduced its range to that portion of the Snake River below Hells Canyon Dam (Simpson and Wallace, 1982).

In determining the viability of a rainbow trout population, an assessment is needed of the quantity and quality of four types of habitat: spawning, rearing, adult, and overwintering habitats. Deficiencies in any one of the four habitat types will limit a trout population (Behnke, 1992). Trout populations may also be limited by food availability (Filbert and Hawkins, 1995). Identifying habitat bottlenecks and the viability of trout populations, however, is difficult without an abundance of hydrologic-, habitat-, and population-related data (Stalnaker et al., 1995).

Optimum riverine habitat for rainbow trout is characterized by: (1) clear, cold water; (2) a silt-free rocky substrate in riffle-run areas; (3) an approximate 1:1 pool-to-riffle ratio that includes slow, deep water; (4) abundant in-stream and stable stream-bank cover; and (5) relatively stable water flows and temperatures (Raleigh et al., 1984). Some information is available for rainbow trout habitat in the Middle Snake River and closely associated tributaries, but it is insufficient to clearly identify the main factors limiting the wild rainbow trout population. In absence of this information, measures of stress to rainbow trout were obtained from habitat suitability indices developed by the U.S. Fish and Wildlife Service (Anglin et al., 1992). Our habitat suitability analysis (i.e., index of impairment) found that the degree of impairment was generally high for all life stages of rainbow trout in the Middle Snake River.

6.1.1. Spawning Habitat

Rainbow trout, stimulated by rising water temperatures, spawn almost exclusively in streams, normally during the period from January to July (Raleigh et al., 1984). Spawning usually occurs when daily maximum temperatures range from 10°C to 15.5°C (Scott and Crossman, 1973), and female rainbow trout are most productive in waters where temperatures do not exceed 13°C for 6 months prior to spawning (Leitritz and Lewis, 1980).

Spawning habitat for rainbow trout in the mainstem of the Middle Snake River has been adversely affected by sedimentation and high temperatures (Hill, 1991c). Our estimates of the quality of the spawning habitat, developed from habitat suitability indices, show that impairment is moderate to high throughout the Middle Snake River except for small river segments just upstream of Kanaka Rapids, where impairment was low (Figure 5-4).

The only known spawning habitat for rainbow trout in the Middle Snake River is located in tributaries off the main channel. For example, spawning occurs in the lower reach of the Malad River, the Thousand Springs Complex, and other short-run springs (Partridge and Corsi, 1993; Lukas et al., 1995). Habitat availability studies by Lukas et al. (1995) found that an estimated 8.5% of the Thousand Springs outfall channels were suitable for spawning. This level exceeds the 5% minimum needed to sustain trout populations (Raleigh et al., 1984). However, because the outflow channels enter the Snake River (the Upper Salmon Falls Reservoir) where no spawning habitat is known to exist, this spawning is not likely enough to sustain populations in both water bodies. Spawning habitat studies are not available for the lower Malad River, but it is likely that fish produced in this area also migrate to the Snake River.

The spawning habitat in the tributaries has also been adversely affected by land and water use decisions. Bell (1980) sampled several springs between Twin Falls and King Hill and found good to excellent populations of wild rainbow trout, which was evidence of natural production in these small tributaries. Subsequently, two of these springs (Lower White and Briggs Springs) were developed for fish culture operations and all rainbow trout habitat was lost (F. Partridge, personal communication, January 6, 1999). It is unknown if similar land use decisions are adversely affecting the remaining springs.

6.1.2. Rearing Habitat

During the first months after hatching, trout need rearing habitat with protective cover and shallow water with low velocity. Cover in the form of aquatic vegetation, debris piles, and interstices between rocks is critical (Raleigh et al., 1984). Some streams may have too much spawning and rearing habitat, resulting in excessive recruitment to populations (Behnke, 1992). However, this does not appear to be the case in the Middle Snake River, where the only identified spawning and rearing habitat occurs in spring seeps, side channels, and small tributaries.

A stable stream flow appears to be important for successful rearing conditions. Year-class strength in a native population of rainbow trout in the Spokane River was positively correlated with spring water flows. A high, relatively constant flow between April 1 and June 25 produced a strong year class, whereas a fluctuating low flow during this time produced a weak year class (Underwood and Bennett, 1992). However, excessively high flows during this time can adversely affect recruitment for trout populations (Binns and Eiserman, 1979).

From 1978 to 1998, the Idaho Department of Fish and Game planted just over 1.5 million fingerling rainbow trout between Glenns Ferry and Milner Dam (records provided by S. Clark, personal communication, January 7, 1999). Most (77%) of these fish were planted between Lower Salmon Falls Dam and Shoshone Falls. According to the Idaho Department of Fish and Game (F. Partridge, personal communication, February 23, 1999), the survival rate for the fingerlings is believed to be dependent on flow. The highest survival was believed to occur during low-flow years, but field studies were not conducted to quantify or qualify the rearing habitat available to these fish.

Lukas et al. (1995) estimated fry and juvenile habitat at 3.5% and 9.1% of the outflow channels, respectively, in the Thousand Springs Complex, but believed there was additional unsurveyed rearing habitat in adjacent small seep springs. Similar studies are not available for the lower Malad River, which also may be an important rearing area for rainbow trout, or for the main stem of the Snake River. We have shown, using habitat suitability indices, that the quality of the rearing habitat for fry and juvenile rainbow trout in the Middle Snake River is highly impaired (Figure 5-4).

6.1.3. Adult Habitat

Adult rainbow trout inhabiting lotic environments typically live at water depths of 0.3 m or greater in areas where slow (0-0.1 m/sec) water for resting is juxtaposed with faster water carrying food and where protective cover is provided by boulders, logs, overhanging vegetation, and undercut banks (Behnke, 1992). In the Middle Snake River, cover for adult fish is limited as streamside vegetation, overhanging banks, and woody debris are not commonly found in this area. In absence of these features, adult rainbow trout use deep pools for cover during the day (Hill, 1991c).

Raleigh et al. (1984) found a definite relationship between the annual flow regime and the quality of trout habitat. The most critical period is usually from late summer to winter, when the lowest flows occur. A stable base flow during this time that is at least 50% of the average annual daily flow is considered excellent for maintaining quality trout habitat, a flow of 25% to 50% is considered fair, and a flow of < 25% is poor. An inspection of the USGS historical streamflow daily graphs from 1970 to 1997 shows that the annual low flow for the King Hill station (No. 13154500) may occur in any month except November or December. However, low flows usually occur from July to September. Information is not available to determine how these low flows compare against the average annual daily flows used by Raleigh et al. (1984).

Adult habitat usually limits the biomass for resident trout in most streams (Behnke, 1992). A determination that adult habitat limits rainbow trout in the Middle Snake River is loosely supported by the fact that rainbow are present in low numbers. Warm-water species, particularly catastomid suckers, are the most common fish species and make up more than 90% of the fish biomass in the Middle Snake River (Lukens, 1982; Dey and Minshall, 1992; Maret, 1997). Our estimates of the quality of the habitat for adult rainbow trout in this reach indicate that it is highly impaired, except for small river segments with low impairment between Rock Creek and Upper Salmon Falls Dam and below Bliss Dam (Figure 5-4).

Lukas et al. (1995) found that a large proportion (68.5%) of the Thousand Springs outfall channels, an area with stable year-round flows, was suitable adult rainbow trout habitat; however, few large fish were present. This was attributed to recreational fishing harvest and migration of adult fish into Upper Salmon Falls Reservoir. Similar surveys are unavailable for the lower Malad River or for the Middle Snake River.

6.1.4. Overwintering Habitat

Overwintering habitat is very important to fish whose survival is related to the amount of deep water with low current and protective cover, such as that occurring in deep pools with large boulders and large woody debris or other types of cover (Bjornn, 1971).

The amount and quality of overwintering habitat for rainbow trout in the Middle Snake River is unknown.

6.1.5. Discussion

Introducing hatchery-reared rainbow trout into the Middle Snake River may have varied effects on the wild rainbow in this area. Vincent (1987) found that stocking catchable-sized hatchery rainbow trout depressed the abundance of wild rainbow and brown trout (*Salmo trutta*) in the Madison River, Montana. Native trout biomass increased after stocking ceased, and the relative degrees of recovery indicated the wild rainbow trout were more negatively affected than wild brown trout. In the Middle Snake River, because wild fish are at low levels, these negative effects may not occur or may be minimal. In the adjacent short-run springs outlets and channels and small tributaries, however, these adverse effects may occur.

Hybridization of native fish is another area of concern when stocking hatchery fish. Whenever hatchery rainbow trout are stocked outside their native range, they almost always hybridize with native fish (Behnke, 1992). Since 1978, at least eight different stocks of rainbow and one rainbow × cutthroat cross have been planted in the Snake River between Glenns Ferry and Milner Dam (S. Clark, personal communication, January 7, 1999.). Given this history, the wild rainbow trout residing in the Middle Snake River likely has a mixed genotype and probably a low potential for survival in the conditions found in this reach. Even in a similar system (Spokane River) that supports native, wild rainbow trout, annual mortality can approach 75% of the population (Underwood and Bennett, 1992).

6.2. MOUNTAIN WHITEFISH *(Prosopium williamsoni)*

The mountain whitefish is native to cold-water rivers and lakes in western North America, both east and west of the Continental Divide (Scott and Crossman, 1973; Sigler and Sigler, 1987), and is widely distributed throughout all of Idaho (Simpson and Wallace, 1982). Unlike more popular game fish, their geographic distribution has changed little over the past century (Rogers et al., 1996). Mountain whitefish are found at elevations ranging from 1,370 to 2,225 m in Utah's Logan River (Sigler, 1951), and above approximately 1,370 m in other parts of its range in Utah, Nevada, and California, apparently because of high water temperatures at lower elevations (McAfee, 1966). As a rule, however, they are not found in small mountain tributaries (Brown, 1952), where Gard and Flittner (1974) believe their upstream distribution is limited by an increased stream gradient and change of substrate from silt-gravel to rubble-gravel. In Western Canada, *P. williamsoni* usually dominates fish communities in mountain lakes at elevations of about 1,400 to 1,950 m when the lake has a large outlet, i.e., stream orders 4-5 (Donald, 1987).

On the basis of fish surveys conducted since 1980, the mountain whitefish population in the Middle Snake River is at a low level. Estimates of the size of the pre-impoundment population of the mountain whitefish in this reach are not available, but as late as 1953-54 "a substantial run" of mountain whitefish used the fishway at Lower Salmon Falls Dam (Irving and Cuplin, 1956). Since 1980, small numbers of fish have been observed in the Bliss reach (Cochnauer, 1981; Maret, 1997), in the Lower Salmon Falls reach (Cochnauer, 1980b), at several locations in the river between Lower Salmon Falls Dam and Shoshone Falls (Dey and Minshall, 1992; Partridge and Warren, 1994), and in the following tributaries to the Middle Snake River: Malad and Big Wood Rivers (Partridge and Corsi, 1993), Little Wood River (Maret, 1997), Thousand Springs outfall channels (Lukas et al., 1995), and Crystal Springs (Hill, 1991a).

Growth rates for the mountain whitefish vary considerably, and fish are smaller for any given age with increasing altitude (McHugh, 1942). Maximum growth rates occur over the first two growing seasons (Pettit and Wallace, 1975) and this fish usually becomes sexually mature from age 2 to 4 years (Brown, 1952; Thompson and Davies, 1976). Female whitefish mature later than the males in a lake environment (Hagen, 1956). The average lifespan of this species is 7 or 8 years (Sigler and Sigler, 1987), although it can live at least 18 years (McAfee, 1966). Seven-year old fish range in length and weight from 307 to 387 mm and from 475 to 890 g, respectively, while the ranges for 8-year-old fish are 330 to 410 mm and 501 to 944 g (Scott, 1960; Pettit and Wallace, 1975; Thompson and Davies, 1976). The largest mountain whitefish on record, which weighed 2,665 g and was 57 cm long, was caught in Island Park Reservoir, Idaho (F. Partridge, personal communication, December 28, 1998). The density of stream populations of mountain whitefish varies greatly by site and season, but may reach a maximum of about 3,400 fish/ha (Wydoski and Helm, 1980).

6.2.1. Loss and Alteration of Lotic Habitat

Dam construction and operation affect riverine environments by reducing the amount of lotic habitat and causing the fluctuation of water levels above and below the dam. There has been a 37% loss of the free-flowing habitat in the Middle Snake River (Cochnauer, 1983). The fluctuation of water levels of impoundments, reservoirs, and tailwaters are both seasonal and diurnal in nature. Of these, the greatest change in water level in the Middle Snake River occurs during diurnal fluctuations in the tailwaters of a dam (Irving and Cuplin, 1956).

The loss of lotic habitat may seriously affect spawning, rearing, and overwintering habitat used by the mountain whitefish (Northcote and Ennis, 1994). Fleming and Smith (1988, in their Figures 10-13) document the reduction in abundance of native cold-water fishes, including the mountain whitefish, and the increase of “coarse” fish (e.g., catostomids and cyprinids) in reservoirs 2 to 42 years after construction in the Upper Columbia River (British Columbia). In contrast, Nelson (1965) observed no decreased growth and little change in distribution of mountain whitefish populations residing in a complex of four reservoirs on the Kananaskis River (Alberta) over 25 years (1936-61) in conjunction with hydroelectric development. Since 1936, however, no appreciable change in water temperature has occurred in three of the four reservoirs. The low water temperatures in the Kananaskis River likely contributed to the stability of the mountain whitefish populations following reservoir development.

6.2.2. Effects on Movement

Upon emergence, the mountain whitefish fry, having a weak swimming ability, drift passively downstream until suitable holding water is encountered. They inhabit shallow areas 5 to 20 cm deep from June to August (Pettit and Wallace, 1975; Davies and Thompson, 1976). Whitefish fry show strong positive phototaxis (Liebelt, 1970), which probably accounts for their selection of shallow backwaters and stream margins. After reaching about 5.5 cm in length, they inhabit low-velocity areas of the river margin having gravel, sand, or mud substrates. Schooling of juvenile mountain whitefish occurs at age 7 months in river and reservoir populations in the Kananaskis River (Alberta) (Nelson, 1965). Yearlings also undergo seasonal migrations between feeding and overwintering habitats, but these probably do not exceed a few kilometers (Northcote and Ennis, 1994).

Movement by adults appears to be complex and includes both nonmigratory and migratory behavior, and some fish exhibit a homing behavior when displaced (Erickson, 1966; Liebelt, 1970). For example, there is a nonmigratory population in the Sheep River watershed in Alberta (Canada), but the majority have annual movement patterns including spawning migrations of > 60 km (Davies and Thompson, 1976, Northcote and Ennis, 1994). Other mountain whitefish populations inhabiting rivers in Utah (Sigler, 1951; Wydoski and Helm, 1980), Montana (Brown, 1952), and in the southern part of their range in North America (McAfee, 1966) did not appear to travel long distances to spawn. Pettit and Wallace (1975) found that mountain whitefish movement usually slows and apparently ceases during the summer, but they did observe movements of > 80 km by some adults from late May to July in the watershed of the North Fork Clearwater River in Idaho. Erickson (1966) observed some trophic movements by whitefish out of the Snake River (Wyoming) into tributary streams during the spring and summer.

Of the three dams located in the Middle Snake River, the Bliss Dam is a barrier to upstream movement by mountain whitefish because it lacks a fish ladder. The other two dams at Lower and Upper Salmon Falls have fishways, but they generally are not in operation (F. Partridge, personal communication, December 28, 1998). Irving and Cuplin (1956) observed a substantial run of mountain whitefish using the fishway over Lower Salmon Falls dam; however, fewer fish used the fishway over the Upper Salmon Falls Dam, owing in part to the poor location of its lower entrance. Irving and Cuplin believed that the dams in the Middle Snake River, with or without fishways, did not present much of an obstacle to the downstream movement of fish. They noted that nearly 50% (197 fish) of the tagged rainbow (*Oncorhynchus mykiss*) planted above the Lower Salmon Falls Dam were recovered below the dam. The mortality of rainbow, or mountain whitefish, passing through hydroelectric installations was not determined.

Unrestricted seasonal movements by mountain whitefish to feeding and overwintering habitats and to spawning areas by adults is no longer possible in the Middle Snake River. The extent of the adverse effects on these movement caused by the construction and operation of dams in this reach, however, has not been determined.

6.2.3. Effects on Spawning Activities

Mountain whitefish usually spawn from October to December (McAfee, 1966; Sigler and Sigler, 1987), but may spawn earlier or later depending on the latitude and water temperature. In the northern part of their range or at higher elevations, whitefish spawn from September to October (Thompson and Davis, 1976), and lake populations may spawn from September to February (Hagen, 1956; McPhail and Lindsey, 1970). These fish spawn in riffle areas of streams where sediment particle size may range from fine gravel to coarse rubble (Brown, 1952). These fish do not construct redds, and spawning typically occurs nocturnally when a small number of fish move into tributary streams from a large river, or onto the margin of a lake. In riverine environments, the fish concentrate in shallow water with depths ranging from 13 to 122 cm and currents ranging from 63 to 155 cm/sec (Brown, 1952; Thompson and Davies, 1976). In Phelps Lake (Wyoming), mountain whitefish spawned at water depths ranging from 6 to 12 m in areas with fine or coarse rubble substrate (Hagen, 1956). The eggs are adhesive when released and stick to the bottom substrate (Sigler and Sigler, 1987).

The thermal requirements for gonadal development of mountain whitefish are unknown, but Ihnat (1981) believes exposure to unnaturally high water temperatures for an extended time during the fall could be detrimental to gamete maturation. Mountain whitefish spawn at water temperatures ranging from 0°C to 9°C, but usually in the 3°C to 5°C range (Northcote and Ennis, 1994). Brown (1952) noted that they do not spawn until the water temperature decreases below 5.5°C and that peak spawning activity at high elevations in the Madison River (Montana) occurred with temperatures just over 2°C.

The upper optimum water temperature for successful egg development is 6°C. Some hatching occurs at water temperatures ranging from 9°C to 11°C, but there are high levels of alevin mortality and abnormality. In a laboratory study, egg mortality was 98% to 100% at 11°C, and at 12°C, all embryos died within 2 weeks (Rajugopal, 1979). In a hatchery, Thompson and Davies (1976) noted that 61 days were required to hatch 95% of fertilized eggs at a temperature of 7.5°C. The last eggs hatched 71 days after fertilization. At 7.2°C, Stalnaker and Gresswell (1974) found that the incubation period for mountain whitefish eggs raised in a laboratory ranged from 52 to 76 days. Optimum temperatures for growth of whitefish fry ranges from 9°C to 12°C (Stalnaker and Gresswell, 1974; Rajugopal, 1979). Juveniles and adults of this species have been collected in the summer in areas with water temperatures of 11°C to 20°C (Ihnat and Bulkley, 1984).

Simulations of water temperatures at two locations (Rkm 896; RM 556.4 and Rkm 956; RM 593.7) in the main stem of the Middle Snake River were evaluated from 1970 to 1994 to determine if they were favorable for mountain whitefish spawning and incubation. These locations were chosen because they are below and above the Thousand Springs complex, which affects the water temperature in the main stem of the Snake River. Favorable spawning conditions were identified as 75 days with a water temperature of <7°C during a possible spawning-incubation window extending from September to February. Modeling results for Rkm 896 (RM 556.4; 5 km below Bliss Dam) indicate that favorable spawning and incubation did not occur in the Bliss reach from 1970 to 1994. In this reach, the number of consecutive days during the potential spawning-incubation window with water temperatures <7°C ranged from 5 (1977) to 45 (1985). Modeling results for Rkm 956 (RM 593.7; near Empire Rapids) also found that a 75-day spawning-incubation window did not occur in this reach from 1970 to 1994. However, water temperatures in this reach were somewhat lower than those in the Bliss reach. In 13 of 25 years, water temperatures <7°C occurred from 45 to 55 consecutive days from September to February.

Impairment of spawning habitat for the mountain whitefish was also estimated in this assessment with habitat suitability indices developed by the U.S. Fish and Wildlife Service (Anglin et al., 1992). This analysis showed that the spawning habitat available to this species is highly impaired in the Middle Snake River (Figure 5-5).

6.2.4. Loss and Alteration of Rearing Areas

Fry and yearling mountain whitefish are found along the shore in water depths of only 5 to 20 cm in small, well-protected pockets created by rubble or boulders and in backwater areas connected to the main stream (Brown, 1952). In tailwaters below dams, the water levels fluctuate diurnally and disrupt rearing by mountain whitefish by pushing them into deeper water where they are more susceptible to predation. Stranding of small fish when the water level drops rapidly is also a concern. Depending on the season, water levels varied from 1.5 to 2 m in the tailwaters of the Lower Salmon Falls Dam, and from 1 to 1.5 m at 22 km and from 0.5 to 1 m at 56 km below Bliss Dam (Irving and Cuplin, 1956). These fluctuations exposed shallow water areas to drying and, during the winter, freezing temperatures. The average surface area of the zone of fluctuation in four typical sections of the tailwaters below these dams was 18% of the surface area as measured during high diurnal flows (Irving and Cuplin, 1956).

The early life stages of the mountain whitefish appear to be able to withstand a reduction in dissolved oxygen. Siefert et al. (1974) found that a reduction in dissolved oxygen concentration to 50% saturation at 4°C and 7°C did not affect the survival of young mountain whitefish, but did delay hatching and early larval growth. However, Siefert believed a dissolved oxygen reduction to 35% saturation would be harmful to survival and a drop to 25% would be lethal to most early stages of the whitefish.

Impairment of rearing habitat for mountain whitefish fry and juveniles was also estimated in this assessment with habitat suitability indices developed by the U.S. Fish and Wildlife Service (Anglin et al., 1992). This analysis showed that the impairment of this habitat was moderate to high in the Middle Snake River (Figure-5-5).

In studying the microhabitat of the mountain whitefish in the Fisher River (Montana), DosSantos (1985, Table 7) observed that as this species increases in size, it occupies deeper, slower waters. The habitats for these fish as described by average facing velocity, 0.6 depth velocity, and water depths were 3.44 cm/s, 16.06 cm/s, and 0.695 m for small fish (10 to 19.9 cm in total length, TL); 1.86 cm/s, 13.47 cm/s, and 0.988 m for medium fish (20 to 27.7 cm TL); and 1.13 cm/s, 9.48 cm/s, and 1.25 m for large fish (>27.7 cm TL), respectively. During the season of least flow in the Logan River (Utah), adults could be found in pools at least 4.8 m wide and having 0.9 to 1.2 m of water depth (Sigler, 1951).

Similar to the evaluation of spawning habitat, rearing habitat for mountain whitefish in the Middle Snake River was moderately to highly impaired under the previously mentioned habitat suitability indices (Figure 5-5).

6.2.5. Effects Due To an Altered Food Source and Prey Base

Mountain whitefish feed primarily on immature forms of bottom-dwelling aquatic insects such as Diptera (true flies and midges), Trichoptera (caddisflies), Ephemeroptera (mayflies), and Plecoptera (stoneflies) (Wydoski and Whitney, 1979). The general morphology of the mountain whitefish with its subterminal mouth suggests it is adapted to feeding on or near the bottom (Pontius and Parker, 1973). Underwater observations of bottom feeding by DosSantos (1985) confirm that this fish actively forages by overturning rocks and plowing through substrates of small particle sizes. In other underwater observations, Thompson and Davies (1976) noted feeding on drift organisms within about 2 to 10 cm of the bottom. Large mountain whitefish can be more generalized feeders and utilize a greater range of prey than do smaller fish. For example, gastropods, small crayfish, leeches, dragonfly nymphs, the amphipod *Gammarus*, young fish, whitefish eggs, and eggs of other species are sometimes eaten by adults (McHugh, 1940; Sigler, 1951; McAfee, 1966; DosSantos, 1985). When bottom fauna are scarce, this fish will eat midwater plankton and surface insects, but under these conditions, the fish is usually less abundant (Godfrey, 1955).

The larvae of midges and caddisflies and the nymphs of mayflies and stoneflies represent very important food items for juvenile and adult mountain whitefish (Pontius and Parker, 1973; Thompson and Davies, 1976; Sigler and Sigler, 1987). A summary of the diet of mountain whitefish from field and laboratory studies conducted from 1936 to 1981 can be found in Table 6-1. With some exceptions, adult fish eat the same aquatic insects in about the same proportion in Montana (Yellowstone, Gallatin, and Kootenai Rivers), Wyoming (Upper Snake River), Alberta (Sheep River), and likely throughout the range of this species. The exceptions occur when whitefish feed heavily on a large emergence of a particular invertebrate or on terrestrial organisms when their aquatic food base is at a low level (Table 6-1). For example, freshwater opossum shrimp, *Mysis relicta,* were the most common food item found in the stomachs of mountain whitefish during a January 1991 study in Kootenay Lake and Brilliant Reservoir (an impoundment on the Kootenay River), and in two areas on the Columbia River in British Columbia (Boyle et al., 1992).

Generally, dipterans and caddisflies are more commonly eaten by mountain whitefish, but mayflies and stoneflies are important food sources at some locations and during some seasons. Overall, chironomids appear to be the most important food item for both juvenile and adult fish; at times they make up nearly 100% of the diet of small fish (Pontius and Parker, 1973; Stalnaker and Gresswell, 1974).

Stream flow regulation at hydroelectric dams can alter the downstream thermal regime and the amount of drifting particulate organic matter. Hauer and Stanford (1982) observed these phenomena and the resultant change in caddisfly community structure in the Flathead River below Hungry Horse Dam. In describing the distribution and abundance of macroinvertebrates in the vicinity of Kanaka, Empire, and Boulder Rapids in the Middle Snake River, Hill (1991b) found that the quality and quantity of the food base for trout were limited. The majority of aquatic insects present were tolerant of organic enrichment, indicative of deteriorated water quality conditions (increased nutrients and sedimentation and depressed dissolved oxygen). As whitefish and trout eat the same macroinvertebrates (McHugh, 1940; Laakso, 1951; Godfrey, 1955; Ellison, 1980; DosSantos, 1985), the prey base for the mountain whitefish is also probably being adversely affected by the deteriorated water quality conditions in the Middle Snake River.

Irving and Cuplin (1956) also found that production of aquatic invertebrates, the primary source of food for whitefish, was adversely affected by diurnal fluctuation of water levels in the tailwaters of the Lower Salmon Falls and Bliss Dams. They found that food production for cold-water game fish was virtually absent within the zone of fluctuation in the tailwaters of these

Table 6-1. Summary of studies on the diet of the mountain whitefish, 1936 to 1981

Location	Date	Fish	Approx. length (cm)	Appox. % volume of occurrence of food in stomach					Method	Reference
				Diptera	Trichoptera	Ephemer.	Plecoptera	Other		
Four lakes & rivers in s. BC & Alberta	1936-38	46	age 1 fish	40-76	4	3-24	6-10	32 (cladocerans)	% occur. by volume	McHugh, 1940, Table 1
Yellowstone R. , MT	Jun-Jul 47	35	3-5	100	8-16	52-77	15	-	% stomachs w/ organism	Lasskso, 1951, Table 1
Yellowstone R. , MT	Fall 47-Sum 48	261	> 18	Tr-61	5-44	5-53	9-39	4-44 (fish eggs)	% total volume	Laakso, 1951, Table 2
Gallatin R., MT	Fall 47-Sum 48	69	> 18	3-33	6-54	1-5	11-48	3 / 9 (fish/beetles)	% total volume	Laakso, 1951, Table 3
Logan R., UT	Aug 48-Jun 49	78	19-47	32	43	9	4	2 (worms & beetles)	% total volume	Sigler, 1951, Table 8
Yellowstone Watershed, MT	Mar 70-May 70	91	1.2-3	97	0	2	4	4 (copopods)	% total freq. occurrence	Liebelt, 1970, Table IX
Upper Snake R., WY	May-Oct 70	40	< 23	99-100	0-1	0-Tr	0	0	% total no. of food organisms	Pontius & Parker, 1973, Table 1
Upper Snake R., WY	May-Oct 70	40	23-33	82-94	4-17	Tr-4	Tr	0.5 (beetles)	% total no. of food organisms	Pontius & Parker, 1973, Table 1
Upper Snake R., WY	May-Oct 70	40	> 33	37-63	10-54	Tr-30	Tr	10 (beetles)	% total no. of food organisms	Pontius & Parker, 1973, Table 1
Logan R., UT	Apr 70-Mar 71	238	1.2-11 (age 0 fish)	94	0.5	3	Tr	2 (beetles)	% total no. of food organisms	Stanlnaker & Gresswell, 1974, Table 5
Sheep R., Alberta	Jul-Oct 73	60	< 20	67-87	1-7	7-13	0-2	9 (terrestrial)	% total no. of food organisms	Thompson & Davies, 1976, Table 5
Sheep R., Alberta	Jul-Oct 73	114	20-35	51-91	3-13	2-28	Tr-4	37 (terrestrial)	% total no. of food organisms	Thompson & Davies, 1976, Table 5
Sheep R., Alberta	Jul-Oct 73	16	> 35	7	12	30	6	40 (terrestrial)	% total no. of food organisms	Thompson & Davies, 1976, Table 5
Teton R., ID	Oct 73	75	11-29	57	18	0	0	12 / 6 (beetles/snails)	% of total volume	Overton et al., 1978 Figure 1
L. Walker R., CA	July 73	64	< 33	26	6	61	1	6	% total no. of food organisms	Ellison, 1980, Table 1
Kootenai, R., MT	Jun 80-May 81	174	10-20	50-78	1-30	4-46	0-1	2	% of total volume	Dos Santos, 1985, Table 1
Kootenai, R., MT	Jun 80-May 81	83	20-28	9-78	4-44	2-48	0-Tr	2	% of total volume	Dos Santos, 1985, Table 1
Kootenai, R., MT	Jun 80-May 81	83	> 28	6-90	6-39	4-26	0-Tr	59 (snails)	% of total volume	Dos Santos, 1985, Table 1

dams. The loss was attributed to the diurnal drying or freezing of this shallow zone of fluctuation, which made up about 20% of the food production area.

During an April-May 1997 macroinvertebrate survey in the 42-km (26 mile) free-flowing reach of the Snake River below C. J. Strike Dam, Cazier (1997b) found the number of species and relative densities were not significantly different between the fluctuation (shoreline area < 2 m deep) zone and the main river.

6.2.6. Discussion

High water temperatures limit the mountain whitefish to elevations above 1,370 m throughout much of its range (McAfee, 1966; Sigler, 1951). Even though the elevation of the Middle Snake River extends from approximately 762 m at King Hill to 1,260 m at Milner Dam, this species appears to have been able to live and reproduce in the Middle Snake River and its nearby tributaries prior to the impoundment of the Snake River. With the increased water temperatures caused by flow depletion for power generation and agricultural irrigation, the river has become an undesirable environment for this cold-water species. This is borne out by our modeling, which shows that water temperatures needed for spawning and incubation are essentially absent; and that water temperatures from mid-April to November on average exceed the maximum optimum temperature, 12°C (Rajagopal, 1979), for growth of mountain whitefish fry each year. Our estimates of habitat impairment using habitat suitability indices also show that most spawning, rearing, and adult habitats available to this species in the Middle Snake River are undesirable.

Competition with other salmonid species for any available habitat left in the Middle Snake River does not appear to be a factor limiting mountain whitefish populations. Across their range in North America, mountain whitefish coexist and may compete with several other fish species. This competition does not appear to have adversely affected mountain whitefish populations, as this fish has evolved with sympatric fishes and appears to occupy a slightly different niche from its competitors. DosSantos (1985) found that the habitats and food sources of small (<20 cm TL) mountain whitefish and rainbow trout were quite similar. However, in comparing the habitat of adult fish (>27.8 cm TL), DosSantos observed rainbow trout in shallower areas with greater velocities and with substrates with double the boulder composition than where adult mountain whitefish were found.

In comparing the food habits of the mountain whitefish with those of cutthroat trout (*Oncorhynchus clarki*) in the Snake River over 2 years, Erickson (1966) showed that only 37% of the total volume of food utilized by cutthroat was also used by the whitefish. The dominant food (>48% of the total volume) for the cutthroat trout was fish. Ellison (1980) found the diets of mountain whitefish and brook trout (*Salvelinus fontinalis*) to be significantly different. Brook trout fed mainly on drifting terrestrial and aquatic insects, whereas whitefish stomachs contained mostly immature aquatic insects, suggesting a bottom-feeding habit. Within its range, the mountain whitefish typically outnumbers sympatric salmonids (McAfee, 1966; DosSantos, 1985).

Others (e.g., Sigler, 1951; McAfee, 1966) believe the mountain whitefish is an important competitor for food and space with more recreationally valuable trout. Mountain whitefish are also reported to compete with the following nonsalmonids for food: peamouth (*Mylocheilus caurinus*), northern pikeminnow (*Ptychocheilus oregonensis*), yellow perch (*Perca flavescens*), and various suckers (*Catostomus* sp.) (Daily, 1971). None of these investigations, however, demonstrated that either the mountain whitefish or a sympatric fish species was detrimentally affected by competition for food or space.

6.3. WHITE STURGEON *(Acipenser transmontanus)*

White sturgeon are presently depressed in abundance throughout their native range in Idaho, and the now landlocked sturgeon in the Snake River is classified as a State of Idaho Species of Special Concern (Idaho Department of Fish and Game, 1994). These fish have evolved life history characteristics that have allowed them to thrive for centuries in large, dynamic river systems containing diverse habitats with multiple food sources. These characteristics include opportunistic food habits, delayed maturation, longevity, high fecundity, and mobility (Beamesderfer and Farr, 1997). White sturgeon appear to have an innate hypometabolic response to hypoxia by reducing spontaneous swimming activity (Crocker and Cech, 1997), which may increase their survival during prolonged hypoxic conditions.

Unfortunately, many of the adaptations by white sturgeon for living in large rivers are now working against maintaining viable populations in riverine environments altered by dam construction. Between its confluence with the Columbia River and Shoshone Falls, 12 hydroelectric projects on the main stem of the Snake River have changed the river into separate, smaller, and less diverse habitats. In the Middle Snake River, there has been a 37% loss of free-flowing habitat (Cochnauer, 1983), which is a direct result of operating dams for hydroelectric power, flood control, and agricultural purposes. Important white sturgeon spawning, rearing, and feeding areas have been changed and, in some cases, lost as a result. These changes, along with past overharvesting, have resulted in significant decline of white sturgeon populations in the Middle Snake River.

Little historical information is available for Idaho white sturgeon populations, but past harvest and abundance trends are believed to be similar to those in the Columbia River. Unregulated commercial harvest in the late 1800s significantly reduced white sturgeon populations throughout the Columbia River basin. Sturgeon populations recovered during the early to mid-1900s, only to be reduced again by a resurgence of fishing activities and the loss of habitat. The main cause of habitat destruction in the Middle Snake River is directly related to the construction and operation of hydroelectric and irrigation dams. The loss of diversified riverine habitats required by the white sturgeon began with the construction of the Swan Falls Dam in 1901 and continued with Lower Salmon Falls Dam in 1910, Upper Salmon Falls Dam in 1937, and Bliss Dam in 1949.

It is not possible to quantify the actual decline that occurred in the white sturgeon population after the Middle Snake River was dammed because only anecdotal information is available on the size of the population prior to dam construction. For example, in discussing the size of the historic population in this reach, Cochnauer (1983) was only able to determine that this area "was known to contain large numbers of sturgeon," some weighing as much as 307 kg (677 lbs). Presently, the largest number of white sturgeon in the Middle Snake River occurs in C.J. Strike Reservoir; a 1991-93 field study in this area estimated the population of fish > 80 cm and > 160 cm fork length (FL) at 2,554 and 268 sturgeon, respectively (Lepla and Chandler, 1995a). Smaller numbers of sturgeon occur in a free-flowing section of the river below Bliss Dam, Rkm 888 to 898 (RM 552 to 558); between Bliss and Lower Salmon Falls Dams; and between Upper Salmon Falls Dam and Shoshone Falls (Lukens, 1981; Lepla and Chandler, 1995a,b). This remnant population is only a fraction of a larger population that occurred in the preimpounded Middle Snake River.

Patterson et al. (1992) provide detailed records of planting 3,583 white sturgeon from Rkm 844 to 989 in the Snake River in 1989-90. Since 1989, however, roughly 5,200 juvenile sturgeon (mostly age 1 fish) were planted by the Idaho Department of Fish and Game between C.J. Strike Dam and Shoshone Falls (T. Patterson, personal communication, September 9, 1997). Of this total, approximately 960 juvenile sturgeon were planted in the upper reach above Upper Salmon Falls Dam after 1989 (Platts and Pratt, 1992; S. Clark, personal communication, May 9, 1996). Some of these fish have survived and may reproduce in this area, particularly if food supplies and rearing areas are not limited. However, the overall success of the planting operation is unknown.

6.3.1. Loss and Alteration of Lotic Habitat

The loss and alteration of unrestricted flows in the Middle Snake River have caused widespread change and reduction in spawning activities, rearing areas, prey species, and feeding areas for white sturgeon. These adverse effects are discussed using available published studies and the results of simulations of habitat, water quality, and ecological processes (Chapter 6 and Appendix D).

6.3.2. Effects on Movement

Historically, white sturgeon could move freely between the Snake and Columbia Rivers (Cochnauer et al., 1985) and up the Snake River as far as Shoshone Falls (Coon, 1978). The mobility of this fish gave it historic access to diverse spawning, rearing, and feeding habitats in the Snake River. This access was effectively reduced or halted by the construction and operation of irrigation and hydroelectric dams. Of the dams in the Middle Snake River, only the complex of dams and barriers at Lower and Upper Salmon Falls have fishways (Irving and Cuplin, 1956), but these are not adequate for the passage of sturgeon (Cochnauer et al., 1985). The Bliss and C.J. Strike Dams form impassable barriers to at least the upstream movement of sturgeon because they were constructed without any fish passage facilities.

White sturgeon are capable of moving long distances over a short period of time. Galbreath (1985) reported apparently random movement for 1,141 previously tagged white sturgeon in the Lower Columbia River (below Bonneville Dam) from 1976 through 1983; the fastest fish traveled 37 km in 3 days. While studying 29 radio-transmitter-tagged juvenile and adult white sturgeon, TL 83 to 218 cm, in a free-flowing reach in the mid-Columbia River, Haynes et al. (1978) observed movement of 3 to 12 km/week upstream and >15 km/week downstream. The average distance traveled for fish that moved at least 0.8 km from the point of release was 40.2 km (25 miles). Haynes and Gray (1981) continued tracking some of these fish and 19 additional sturgeon (98 to 236 cm TL) that were similarly tagged and observed that long-distance travel appeared to be initiated when water temperature reached 13°C.

The movement of white sturgeon in the free-flowing Hells Canyon reach of the Snake River appears to be more restricted than for Columbia River fish. During a 4-year study in the unimpounded 222 km (138 mi) reach between Hells Canyon and Lower Granite Dams, Coon (1978) observed that 22% (39 of 175 fish) of the small sturgeon (45 to 91.5 cm TL) moved downstream an average of 8.6 km (range 1 to 39 km) from their release site over 1.2 years. The remaining 78% of the small sturgeon and fish larger than 92 cm TL (total of 164 fish) remained within about 15 km of their release sites.

White sturgeon in reservoir systems are also capable of moving considerable distances. After capture and release, sturgeon moved a maximum of 152 km (94 miles) but averaged 8.1 km (5 miles) in the three lowest reservoirs in the Lower Columbia River; the time of travel was not given (North et al., 1993). Fish size did not appear to affect the distance or direction traveled, and 49.9% moved upstream and 50.1% moved downstream. Four percent of the tagged sturgeon moved past a dam (26 fish moved downstream and 1 upstream).

Movement and survival of hatchery-stocked and wild white sturgeon downstream through or over Bliss and C.J. Strike Dams was observed by Lepla and Chandler (1995a, 1997). Observations in the impounded part of the Lower Columbia River (North et al., 1993; Beamesderfer et al., 1995) indicate that white sturgeon rarely use fish passage facilities designed for the passage of salmonids. Avenues for passage more frequently used by this fish are navigation locks, which are not present in the Middle Snake River or downstream, through turbines, or perhaps over spillways.

Access to white sturgeon spawning habitat appears to be limiting recruitment in the Middle Snake River. Spawning habitat for sturgeon populations isolated by dams is limited to that occurring within the reach where the fish live. Because white sturgeon can only move downstream through or over dams, potential spawning areas located upstream of the dams in the Middle Snake River may be underutilized on a long-term basis. This appears to be the case for the white sturgeon in the reaches between Bliss and Upper Salmon Falls Dams and below C.J. Strike Dam. These fish cannot move upstream into known spawning sites or potential spawning sites like Kanaka, Empire, and Boulder Rapids.

Sturgeon living in the Bliss reach have a somewhat different problem reaching spawning habitat. During periods of low flow, the larger sturgeon population in C.J. Strike Reservoir does not appear to be attracted to the numerous spawning sites located upstream between King Hill and Bliss Dam. In 1992-93, fish in this area moved about 6.5 km upstream of the reservoir and spawned in the Grass Hole, but none were observed moving further upstream. The 37 km free-flowing section between Grass Hole and King Hill flows through relatively flat terrain characterized by slow-moving runs with shallow riffles and few deep holes (Lepla and Chandler, 1995a). During low-flow (e.g., < 255 m^3/s; 9,000 cfs) conditions, this reach may restrict the upstream movement of white sturgeon to spawning areas above King Hill.

6.3.3. Effects on Spawning Activities

Spawning by white sturgeon in the impounded Middle Snake River is limited by low numbers of adult spawners in the reaches above Bliss Dam, loss of access to historic spawning areas, loss of high flows, and poor water quality during the spawning season.

In the Columbia River basin, white sturgeon usually spawn in areas of high current velocity over large rocky substrates from February to June (Platts and Pratt, 1992; Parsley et al., 1993). A high discharge rate (flow) appears to stimulate spawning activity. A high flow appeared to cue white sturgeon spawning activity in the tailrace of The Dalles Dam. Water temperature, however, was not an acute spawning cue, unlike higher river discharge, discharge coefficient of variability, and water column velocity (Anders and Beckman, 1995). Mean water column velocities near white sturgeon spawning sites in the Lower Columbia River were 0.8 to 2.8 m/sec, and velocities near the substrate were 0.5 to 2.4 m/sec (Parsley et al., 1993).

The amount of white sturgeon spawning habitat available in the tailraces of four dams on the Lower Columbia River is controlled by the discharge rate and water temperature. The amount (area) of spawning habitat in these tailraces increased as discharge increased, and the spawning period was defined by a range of water temperatures acceptable for spawning (Parsley and Beckman, 1994). White sturgeon in the Columbia River spawn in water temperatures of 10°C to 18°C, with most spawning occurring at 14°C (Parsley et al., 1993). Successful egg incubation is possible within a temperature range of 10°C to 18°C, but best results occur at 14°C to 16°C. Temperatures 18°C to 20°C may cause substantial mortalities during sensitive embryonic stages, and temperatures above 20°C are clearly lethal to white sturgeon embryos (Wang et al., 1985, 1987). In-laboratory growth of white sturgeon weighing <1 g increased significantly from 15°C to 20°C, but was not significantly different from 20°C to 25°C (Cech et al., 1984).

Since at least 1980, recruitment of white sturgeon in the Middle Snake River appears to have been limited by the lack of or low numbers of spawning fish. Lukens (1981) captured only seven fish (ages 4 to 9 years) in the Lower Salmon Falls reach, none between Lower and Upper Salmon Falls Dams, and nine fish (ages 4 to 27 years) in the Shoshone Falls reach in 1980-81. According to Cochnauer et al. (1985), a female white sturgeon may not mature until at least 11 years of age and probably does not spawn but every 3 to 11 years. Eleven-year-old white sturgeon in the Middle Snake River are approximately 125 cm in length. Cochnauer et al. (1985) found only two fish older than 11 years and six fish greater than 125 cm in length between Bliss Dam and Shoshone Falls. A 1993 field survey (Lepla and Chandler, 1995b) in the Lower Salmon Falls reach captured only 38 sturgeon, none greater than 115 cm in length. Lepla and Chandler (1995a) believed that a small number of gravid females, along with poor water quality conditions during the time of egg incubation, were factors limiting recruitment in the Bliss reach in 1991-93.

In areas where adequate numbers of spawning fish exist, the loss of high flows during the spawning season and high water temperatures during embryonic development appear to represent the most important factors limiting successful recruitment of white sturgeon in the Middle Snake River (Lepla and Chandler, 1995a). To evaluate these potential limiting factors, 30 years of flow and water quality conditions were simulated (Chapter 6 and Appendix D). The flow data were developed from actual data; the temperature data are simulated.

Spawning conditions in the upper Bliss reach were assessed using model output developed for the river at Rkm 893 (RM 555). This site is 1.3 km below the known white sturgeon spawning area in Porterfield Hole and is believed to fairly represent conditions in the upper Bliss reach. Spawning problems caused by low flows and high water temperatures at this location are probably indicative of the entire free-flowing area below Bliss Dam. The site at Rkm 893 (RM 555) was modeled because the channel morphology was known at that location.

Potential spawning times for white sturgeon can be identified by evaluating the annual flows and water temperatures for the site at Rkm 893 (RM 555). As discussed above, an increasing water flow appears to cue sturgeon to spawn, and the time frame for spawning can be further defined by a 10°C to 18°C range in water temperature. To assist in identifying potential spawning periods, flows and water temperatures for the site at Rkm 893 (RM 555) are plotted together from March through July, which includes the observed spawning season for this area.

The potential spawning period begins near the end of March, when the water temperature increases above 10°C, and continues for about 3 months until the high flows drop off and water temperatures rise above 18°C (Figure 6-1). Given that 19.5 days are required at 17°C for embryonic

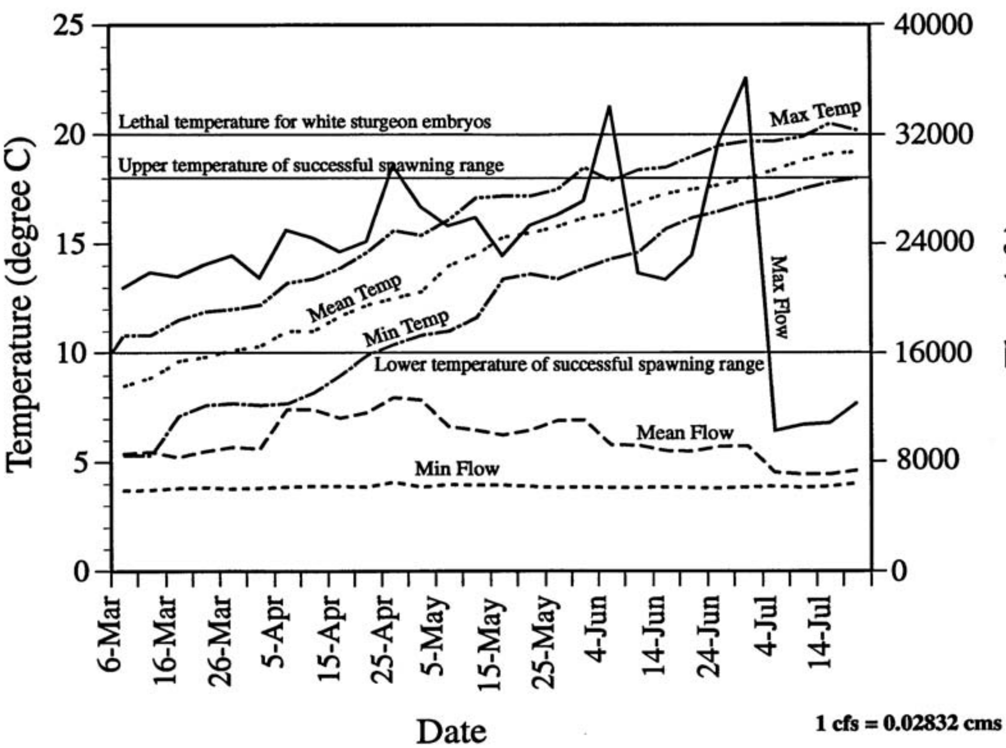

Figure 6-1. Water temperature and flow during the white sturgeon spawning season at Snake River Rkm 893 (RM555).

development to be completed (complete absorption of the yolk) (Wang et al., 1985), the effective spawning period extends from April through the first week of June. Favorable spawning conditions exist when high flows occur during this period. White sturgeon may spawn in the absence of high flow (e.g., see Lepla and Chandler, 1995a, Figures 32 and 33), but these are less than optimum conditions. Optimum spawning conditions are believed to occur when high flows, as represented by the maximum flow in Figure 5, coincide with water temperatures ranging from 10°C to 18°C. Under these conditions, not only is there a longer time period of increasing flows to attract sturgeon for spawning, but the water temperature conditions after spawning are more favorable for the survival of sturgeon eggs and embryos. Simulations of conditions (described in Chapter 6) favorable for white sturgeon spawning and incubation were evaluated. *Favorable* spawning and incubation conditions were identified because the *optimum* conditions are unknown. Using just two variables, flow and water temperature, favorable white sturgeon spawning and incubation periods were identified from 1970 to 1994 by modeling flows >425 m^3/sec (15,000 cfs) with water temperatures ranging from 10°C to 18°C (for spawning) followed by water temperatures <8°C for 20 days (for incubation). The time period modeled each year was from March to July. The modeling results show that during the 25-year period (1970-94) favorable spawning conditions followed by favorable incubation conditions occurred 52% of the time. These conditions occurred in 1970-72, 1974-76, 1978, 1980, and 1982-86. Unfavorable conditions occurred during all other years, including the eight consecutive years from 1987 to 1994. These results may be an underestimate of favorable conditions, as the 425 m^3/sec (15,000 cfs) flow modeled is believed to be too low for favorable spawning conditions. Spawning years 1995-96 were not modeled, but the USGS daily streamflow reports for the King Hill station (No. 13154500) show that flows were more favorable for spawning in both years.

The low level of successful spawning in recent years is further indicated by the predominance of hatchery fish in some areas. For example, 87% (33 fish) captured in 1993 in the Lower Salmon Falls reach and 50% to 87%, depending on capture technique, of fish <80 cm FL in the Bliss reach from 1991 to 1993 were of hatchery origin (Lepla and Chandler, 1995a,b). Fish <80 cm FL would be about 7 to 8 years of age or younger (Cochnauer, 1983).

6.3.4. Predation on Eggs and Larvae

Another factor that may limit white sturgeon recruitment is the loss of eggs and larvae to predators during low-flow conditions. Miller and Beckman (1996) believed that white sturgeon reduce the opportunity for predation from sympatric fishes by broadcasting their eggs in extremely fast-flowing water, which may limit predator access to the spawning and incubation areas. They found white sturgeon eggs in the guts of northern pikeminnow (*Ptychocheilus oregonensis*), largescale sucker

(*Catostomus macrocheilus*), carp (*Cyprinus carpio*), and the prickly sculpin (*Cottus asper*). All except the sculpin can be found in the Middle Snake River, and the largescale sucker may be the most abundant fish in this area (Cochnauer, 1980a,b, 1981; Simpson and Wallace, 1982). The higher velocities (>1 m/sec) used for spawning appear to exclude these predators. Faler et al. (1988) found that northern pikeminnow in the McNary tailrace avoided areas with surface velocities greater than 70 cm/sec. Adult largescale suckers maintain position in water with velocities only up to 1.1 m/sec in laboratory exposures (Dauble, 1986). Carp generally avoid swift current and prefer quiet water in areas with dense vegetation (Wydoski and Whitney, 1979).

Furthermore, the risk of predation should be reduced with increased water flows because at high flows predators must search a greater volume of water for white sturgeon eggs and larvae. Turbid water associated with high flows may also provide protection from visual predators. McCabe and Tracy (1994) theorized that wide dispersal of eggs also allows utilization of more feeding areas and rearing habitats by larval and postlarval white sturgeon and minimizes competition for resources. Poor dispersal during low flows may result in greater loss of developing eggs to fungal and other diseases because clumping of the adhesive eggs would be more likely to occur.

6.3.5. Loss and Alteration of Rearing Areas

The protection and restoration of rearing habitats for white sturgeon do not appear to be as critical as for spawning habitats (Beamesderfer and Farr, 1997). Parsley et al. (1993) captured juvenile white sturgeon in the Lower Columbia River in areas with water depths of 2 to 58 m, mean water column velocities of 1.2 m/sec or less, and substrates of hard clay, mud and silt, sand, gravel, cobble, boulder, and bedrock. Young-of-the-year were found in similar areas. The habitats where these fish were captured indicate a tolerance of, but not necessarily a preference for, a wide range of environmental conditions. In the Lower Columbia River, young-of-the-year and juvenile white sturgeon appeared to have a preference for the thalweg, as that is where Parsley et al. (1993) most often captured them; samples adjacent to the thalweg in shallow water rarely contained sturgeon. The absence of white sturgeon from shallow areas suggests the photonegative behavior observed in the larval stage also occurs in more advanced life stages (Brannon et al., 1985).

Shallow-water areas, however, may represent important rearing habitat for white sturgeon. In the Middle Columbia River, Haynes and Gray (1981) observed diel movements by white sturgeon 98 to 236 cm TL into shallow water nearshore and slough areas in late afternoon and evening and a return to deep water the following morning. These movements were into areas where benthic organisms and smaller fish were more abundant. In the Lower Columbia River, McCabe et al. (1993) believed that juvenile white sturgeon 144 to 724 mm FL fed in water 1 to 6 m deep where a favorite prey item, a tube-dwelling amphipod, reached densities more than 10 times greater than deeper water areas. In the Bliss reach, juvenile sturgeon <129 cm TL used areas with water depths as shallow as 3 m (Lepla and Chandler, 1995a).

Rearing habitat for young-of-the-year and juvenile white sturgeon in the Middle Snake River has not been documented as well as that in the Lower Columbia River, where rearing habitat appears not to be limited (Parsley and Beckman, 1994). Although the Snake River white sturgeon show a tolerance for a wide range of environmental conditions, the riverine-reservoir system in the Middle Snake River has poorer water quality and much lower flows than the Lower Columbia River. These factors may limit the amount of acceptable rearing habitat available to the white sturgeon in the Middle Snake River.

6.3.6. Effects Due to an Altered Food Source and Prey Base

Prior to impoundment of the Snake River, white sturgeon could take advantage of scattered and seasonal food sources by moving between different riverine habitats. They are opportunistic feeders with a wide range of food items including zooplankton, molluscs, amphipods, aquatic larvae, benthic invertebrates, and fish (McCabe et al., 1993; Lepla and Chandler, 1995a). White sturgeon are more predaceous than any other North American sturgeon (Semakula and Larkin, 1968) and can capture and consume large prey (Beamesderfer and Farr, 1997). Seasonal migrations occur in the Lower Columbia River where sturgeon move to feed on eulachon, northern anchovy, American shad, moribund salmonids, amphipods, and other invertebrates (DeVore et al., 1995). This species is also capable of subsisting for long periods when food is not available (Beamesderfer and Farr, 1997).

In the Middle Snake region, use of historic food sources and feeding areas by white sturgeon was greatly impaired by the construction of dams and by the loss of an unrestricted lotic environment. Anadromous fish runs that occurred at different times of the year likely provided very important food sources for the sturgeon. Anadromous species previously entering this area were summer and fall chinook salmon (*Oncorhynchus tshawytscha*), steelhead (*O. mykiss*), and lamprey (*Lampetra tridentata*); some historical evidence indicates that coho salmon (*O. kisutch*) may have also inhabited this area (Dey and Minshall, 1992). This prey base was temporarily stopped from reaching the Middle Snake River about 1901 when Swan Falls Dam was constructed and a fish ladder was not added until about a decade later. This fish passage facility, however, only operates intermittently during high water conditions (Irving and Cuplin, 1956). The anadromous prey base was eliminated entirely in 1952 when C.J. Strike Dam was constructed without any fish passage facilities. Unfortunately, at that time, the only remaining anadromous fish returning to this part of the river was a small run of steelhead (Irving and Cuplin, 1956).

Platts and Pratt (1992) discussed two other historical food sources no longer available to the white sturgeon in the Middle Snake River. Unrestricted river flows transported organic debris in the form of dead fish and incoming terrestrial organisms from upstream locations. The continuous input of organic debris along with incoming gravels and rubble also provided habitat for large mussel and crayfish populations. Both the incoming organic debris and the abundant mussel and crayfish populations were important food sources.

It is ironic that the largest population of white sturgeon at this time in the Middle Snake River resides in C.J. Strike Reservoir, which was created by a dam. Food sources for white sturgeon in the C.J. Strike Reservoir are apparently adequate for maintaining this population. As shown in Table 6-2, the growth rate of sturgeon in this area compares favorably with the unimpounded sections of the Lower Columbia River and the Fraser River in British Columbia. The growth rates of white sturgeon in C.J. Strike Reservoir are a testament to their ability to shift their prey base in a changing environment.

The growth rate of white sturgeon in the Hells Canyon reach of the Snake River was similar to that in C.J. Strike Reservoir. In studying 650 sturgeon (ages estimated to range from 2 to 56 years) in this area, Coon (1978, his Figure 12) found that the growth rate was generally quite rapid up to about 4 years of age (up to 60 cm TL), averaged about 2 cm/yr from ages 4 to 15 years (up to 85 cm TL), increased to about 8 cm/yr from ages 15 to 30 years (up to 210 cm TL), and then decreased to 2 cm/yr from ages 30 to 45 years (up to 240 cm TL). Coon noted that sturgeon growth decreased after the construction of the three dams in Hells Canyon, located upstream of his study area. It appears the existing prey base, which is reduced from historical levels, may not be an important factor limiting growth in the present-day sturgeon population in the Middle Snake River. A return of the anadromous prey base would likely improve the production of white sturgeon, but the return of these fish species

Table 6-2. Total lengths and mean growth rates for white sturgeon in the Middle Snake, main-stem Columbia River in the United States, and Fraser River, Canada

Water body	Total length (cm) 12 yrs	20 yrs	Mean growth rate (cm/yr)	References
C.J. Strike Reservoir	125	180	7.2 from 5 to 25 yrs	Cochnauer et al., 1985
Lower Columbia River	122	183	6.6 from 1 to 21 yrs	Galbreath, 1985
Fraser River[a]	97	142	5.1 from 5 to 25 yrs	Scott and Crossman, 1973

[a]All were female sturgeon and with fork lengths converted to total lengths (TL = FL × 1.110).

is a much larger issue than just providing for adequate passage at the main-stem dams. An assessment of the long-term reproductive survival of each of these anadromous species similar to that being conducted for white sturgeon would be required. Their life histories and environmental requirements are more complex than those of the sturgeon and would require significant repair not only on main-stem Snake River habitats, but on important tributaries as well.

6.3.7. Loss of Genetic Diversity

Genetic diversity is required for the maintenance of long-term health and robustness of any biological community. In reviewing the evolution of animals, Brown (1985) noted that mitochondrial DNA (mtDNA) has been widely used as a genetic marker in population studies because of its higher mutation rate relative to nuclear DNA, generally maternal inheritance, and lack of sexual recombination. Brown et al. (1992) studied mtDNA variation in *A. transmontanus* populations in the Fraser and Columbia Rivers. With regard to Idaho white sturgeon populations, genetic analysis of Kootenai River fish by Setter and Brannon (1990) indicates they are a unique stock and constitute a distinct interbreeding population. These fish have been isolated from the Columbia River for about 10,000 years (Northcote, 1973). An analysis of the genetic diversity of the white sturgeon populations in the Middle Snake River has not been performed.

The three-year study by Brown et al. (1992) suggests that overexploitation of the white sturgeon in the Columbia River during the late 1800s and early 1900s and the subsequent habitat destruction caused by the construction of hydroelectric dams caused a “genetic bottleneck” that reduced mtDNA or genetic diversity in Columbia River white sturgeon. Brown et al. (1992) found a steep reduction in the mtDNA diversity in the landlocked sturgeon populations in the Columbia River. Of six observed genotypes in the river, the upriver population above McNary Dam shares only one genotype with fish found below Bonneville Dam.

At present, information is not available to determine if there is a reduction in genetic diversity in the Snake River white sturgeon populations. However, because this population has experienced very similar overexploitation and habitat destruction due to the construction and operation of dams, and appears to be even more isolated than any of the mainstem Columbia River fish, it is likely that a reduction in mtDNA diversity is occurring.

As discussed by Brown et al. (1992), any reduction in the genetic diversity in sturgeon populations should be of special concern in the conservation of this species, given its long generation time and advanced age at reproductive maturity.

6.3.8. Discussion

The populations of all eight species of sturgeon in North America are presently depleted, threatened, or extinct (Smith, 1990; Birstein, 1993). Recovery efforts for white sturgeon and other sturgeon species in North America have tended to focus on flow augmentation during spawning periods and supplementation of populations through hatchery rearing programs (Hildebrand et al., 1999). These two efforts have been used with some success in the Kootenai River in northern Idaho (Duke et al., 1999), and are planned for the Upper Columbia River in British Columbia (Hildebrand et al., 1999). Beamesderfer and Farr (1997), in a survey of sturgeon and paddlefish in North America, identified protecting and restoring critical habitat, especially for spawning, restoring the natural hydrograph, and improving passage at dams as being the most important for benefiting the conservation, productivity, and diversity of these fish.

Over the past century, white sturgeon in the Middle Snake River have experienced a reduction of their diverse riverine environment and an alteration of the dynamic hydrologic cycle to which they are historically adapted. In some cases, for example in the C.J. Strike Reservoir and in the free-flowing reach from King Hill to Bliss Dam, the white sturgeon has survived. But overall, judging by their reduced and declining population numbers, the white sturgeon is not adapting well to the habitat alteration, unfavorable spawning and rearing conditions, and isolation caused by the dams. These unfavorable environmental conditions are worsened during periods of drought.

With one exception, the habitat suitability analysis showed that unfavorable living conditions (i.e., the highest impairment) for all life stages of the white sturgeon generally occurred above Lower Salmon Falls Dam (Figure 5-6). The exception was that the juvenile and adult stages had the lowest impairment in the reach from Box Canyon downstream to Upper Salmon Falls Dam. In the reaches downstream from Lower Salmon Falls Dam, living conditions were shown to improve, except for the Bliss Reservoir, which had unfavorable living conditions for all life stages (Figure 5-6).

Even though significant restoration of lost habitat for the white sturgeon will not occur without dam removal, it is possible to improve conditions by altering the way the dams are operated in the Middle and Upper Snake River. Past studies have shown that spawning and early embryonic development have been unsuccessful in large part because of poor water quality (Lepla and Chandler, 1995a), which has resulted in depressed natural recruitment. These adverse conditions can be improved if a spring freshet is reestablished with flows large enough to stimulate spawning, and with post-spawning water temperatures low enough (<18°C) to allow for healthy embryonic development other than during a narrow window of time.

Improved recruitment to the white sturgeon population may also be achieved by reducing or avoiding block loading (or load shaping) at the dams during the spawning season. During these loading events there is a sudden increase followed by a sudden decrease in water flows each day. Observations in the Snake River (Chandler and Lepla, 1997) and in the Columbia River (Hildebrand et al., 1999) show that this type of flow management at dams my influence the intensity of spawning and the survival of spawned eggs.

Anglin et al. (1992) used Instream Flow Incremental Methodology (IFIM) to develop habitat vs. flow relationships for white sturgeon in the 241-km section of the Snake River from C.J. Strike Dam to Brownlee Reservoir. Habitat time series analysis (Anglin et al., 1992) shows that a 339.8 m^3/s (12,000 cfs) flow scenario provided the best habitat for adults, juveniles, spawning, and incubation. In general, habitat for these white sturgeon life stages continued to increase with flows up to at least 425-566 m^3/s (15,000 to 20,000 cfs). However, the flows for the scenarios identifying critical habitat for white sturgeon life stages may be low, at least for spawning and incubation. Marcuson et al. (1995) and Paragamian et al. (1997) recommended minimum flows of 991-1,132 m^3/s (35,000 to 40,000 cfs) in their investigations of flows necessary to produce conditions favorable for recruitment of white sturgeon in the Kootenai River.

The freshets would also scour the annual buildup of sediment from deep holes that provide rearing habitat for white sturgeon. Without annual freshets, the bedload transport may gradually fill some of these deep holes, particularly in areas with low gradients. During a 1979-81 study, Lukens (1982) described the lower part of the Bliss reach below King Hill as "relatively flat (0.32 m/km) with few distinguishable shallow riffles, long runs, and thirteen traditional sturgeon fishing holes." Just over 10 years later, Lepla and Chandler (1995a) found few sturgeon (4% of the Bliss reach population or 24 fish) inhabiting this reach. Rearing habitat for sturgeon appears to have been reduced in this area. Anecdotal information from "older local fishermen" indicates that sediment has filled sturgeon holes in the reach below C.J. Strike Dam (Lukens, 1982). Poor water quality during the summer may also have attributed to the low number of fish captured during the 1991-93 surveys in the low-gradient area of the Bliss reach.

7. ANALYSIS OF EXPOSURE AND EFFECTS FOR MACROINVERTEBRATES

7.1. OVERVIEW

Aquatic macroinvertebrates are the primary tool for bioassessment of freshwater ecosystems (Rosenberg and Resh, 1993). A quantitative analysis of macroinvertebrate communities provides a comprehensive assessment of the overall ecological integrity of a system, particularly when combined with information on water chemistry and in-stream habitat conditions (Minshall, 1993). Fish assemblages have been used with noted success as a bioassessment tool in streams and rivers in the midwestern United States (e.g., Karr et al., 1986). In the western United States, however, the use of fish assemblages as a bioassessment tool is constrained by the natural paucity of species, stocking of game fish, and the anadromous life history of many native species (Fore et al., 1996). The sessile nature of macroinvertebrates (relative to fish) makes the macroinvertebrate community of a river a natural integrator of physical and chemical conditions within that system. The bioassessment of freshwater ecosystems in the western United States should focus on macroinvertebrates, although the assessment of fish assemblages still can provide useful information, particularly when focused on fisheries concerns or toxicological endpoints (see Maret et al., 1997; Clark et al., 1998; and Maret, 1999, for examples from the Snake River).

This chapter is presented in two sections. The first section describes the macroinvertebrate community of the Middle Snake River at several locations between Twin Falls and Hagerman, ID. This sampling was conducted by Idaho State University from 1992 through 1994. The second section reviews the status of several of the threatened or endangered molluscs found in the Middle Snake River.

7.2. SAMPLING BY IDAHO STATE UNIVERSITY 1992-1994

During the time of this sampling effort, the riverbed of the Middle Snake River was composed predominantly of soft sediments with occasional large boulders. Only the sampling location immediately downstream of Pillar Falls (Rkm 980.8, RM 613) contained abundant gravel and small cobble-sized substrate (Todd V. Royer, personal observation).

Because of the reduced water velocity, favorable substrate, and elevated concentrations of dissolved nutrients, the Middle Snake River at that time supported dense populations of aquatic macrophytes, primarily *Ceratophyllum demersum*, *Potamogeton pectinatus*, and *P. crispus* (Falter et al., 1995). These macrophytes provided substrate for filamentous green algae, mostly *Cladophora* and *Hydrodictyon* (Falter et al., 1995), which formed thick mats on the surface of the river during the summer months.

The abundant growth of aquatic macrophytes and filamentous algae, together with the high mean concentrations of nitrogen and phosphorus (Table 4-6), indicate that the Middle Snake River was a highly eutrophic system during the years 1992-94. The eutrophic condition also was reflected in the extremely fast rate at which organic material in the river decomposed (Royer and Minshall, 1997). In addition to the eutrophic condition of the river, reduced water velocity due to impoundment allowed greater solar heating of the water. In 1994 (a high-flow year), monitoring of water temperature in the Middle Snake River near the Magic Valley Fish Hatchery (approximately Rkm 944; RM 590) revealed 40 days during July-August in which the mean daily water temperature exceeded 20°C. Furthermore, 35 of the 40 days were consecutive, essentially precluding the existence of cold-water biota from the Middle Snake River.

Mean density of aquatic macroinvertebrates in the Middle Snake River often exceeded 75,000 and occasionally approached 200,000 individuals per m^2 (Figure 7-1). The exception to this occurred in April 1994 when density at all stations was less than 50,000 individuals/m^2. These values of density are high relative to other rivers in southern Idaho. Density of macroinvertebrates in the Owhyee River in 1995 was approximately 12,300 individuals per m^2 (Royer and Minshall, 1996), or about 25% of that in the Middle Snake River. Despite the high density of macroinvertebrates, taxa richness was low, never exceeding a mean value of 15 taxa at any sampling location (Figure 7-2). The greatest values for richness usually were observed between Rkm 931.6 and Rkm 952.5 (RM 579 and RM 592), the most downstream stations examined in the study. In 1993 and 1994, the most upstream location (Rkm 986.3, RM 613) also displayed relatively high taxa richness, although the absolute values always were less than 12 taxa. The lowest values of richness occurred around Rkm 957.4 (RM 595), with mean richness values of 3 to 7 taxa over the duration of the study.

The taxonomic resolution of the study was typically to genus, although some of the most abundant groups (e.g., Chironomidae, Oligochaeta, and Turbellaria) were left at a more coarse resolution. Taxonomic resolution notwithstanding, the macroinvertebrate community of the Middle Snake River during 1992-94 displayed a paucity of taxa. For example, quantitative sampling of the Owhyee River in southern Idaho, using a similar level of taxonomic resolution, revealed 24 taxa in 1995 (Royer and Minshall, 1996), whereas no location sampled in the Middle Snake River exceeded 15 taxa on any date.

In addition to the low diversity of taxa in the Middle Snake River, the macroinvertebrate community was composed primarily of pollution-tolerant and sediment-dwelling taxa (Table 7-1). Typically, the 3 to 4 most common taxa made up 80% to 95% of the benthic macroinvertebrate community. Throughout the Middle Snake River from 1992 to 1994, Oligochaeta, *Hyalella azteca*, Chironomidae, Turbellaria, and *Potamopyrgus antipodarum* were

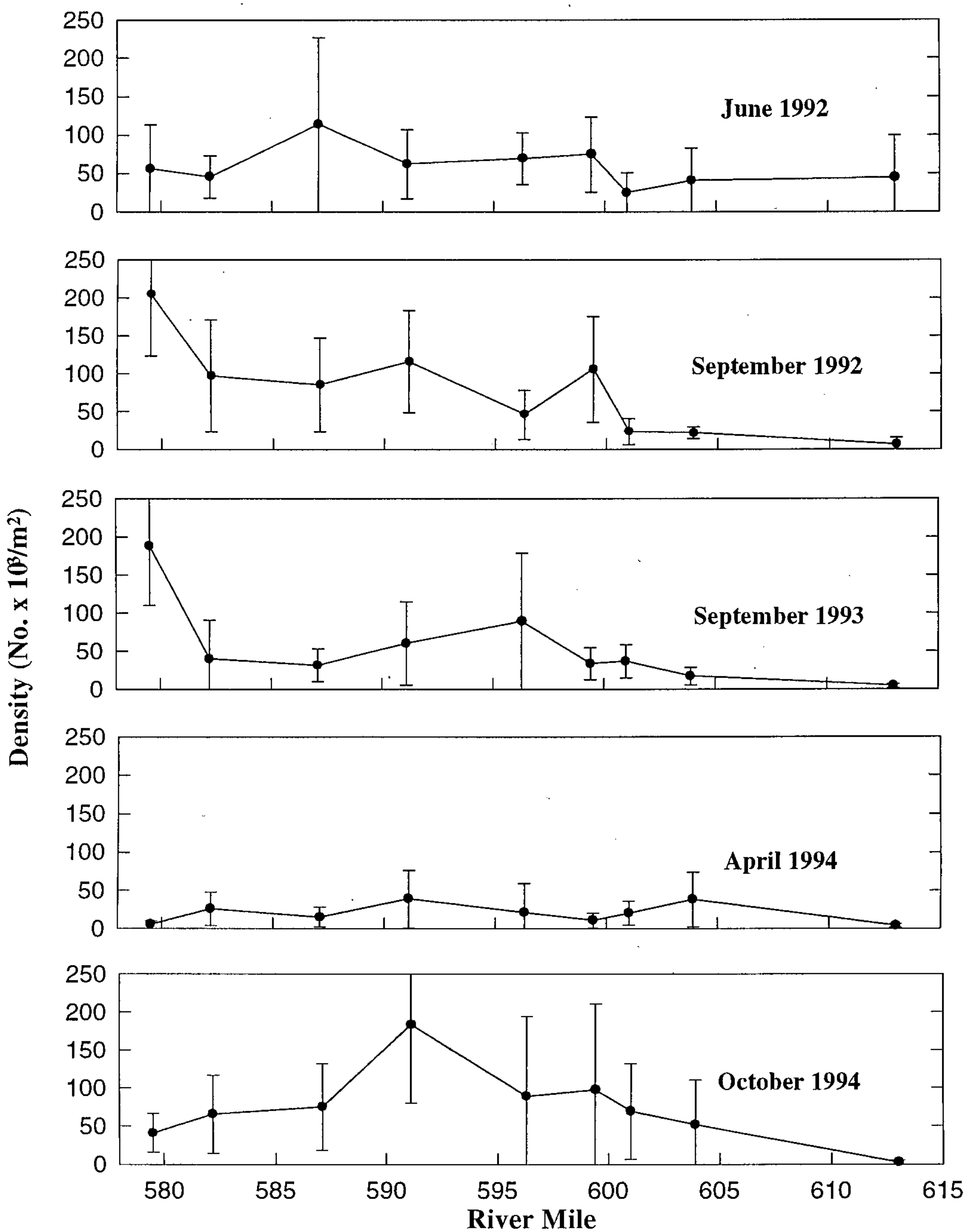

Figure 7-1. Mean density of aquatic macro invertebrates in the Middle Snake River at locations sampled by Idaho State University. Error bars equal +/- SD, n=9.

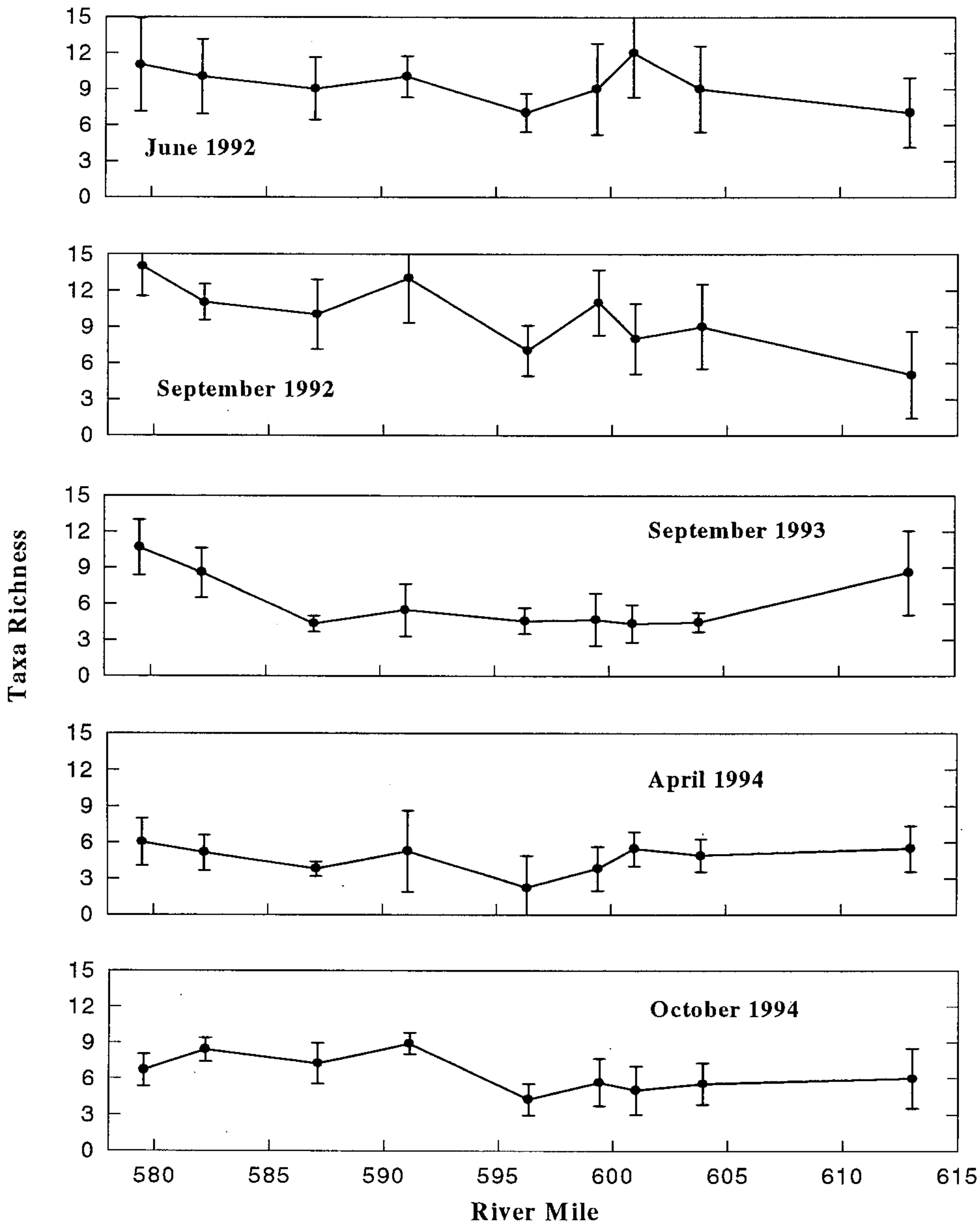

Figure 7-2. Mean taxa richness of aquatic macroinvertebrates in the Middle Snake River at locations sampled by Idaho State University. Error bars equal +/- 1SD, n=9.

Table 7-1. Mean (SD) relative abundance (%) of the ten most common invertebrate taxa in the Middle Snake River on each of the sampling dates. Values for each date calculated from all nine sampling stations (from Royer et al., 1995)

	Jun 1992	Sept 1992	Sept 1993	Apr 1994	Oct 1994
Potamopyrgus antipodarum	38.0 (21.4)	51.1 (27.7)	63.7 (27.1)	31.2 (14.5)	70.0 (19.4)
Oligochaeta	44.5 (16.9)	29.6 (22.6)	21.0 (17.6)	58.1 (13.1)	17.5 (6.5)
Hyalella azteca	2.9 (6.3)	2.9 (3.8)	5.7 (13.2)	1.4 (3.0)	4.6 (10.6)
Chironomidae	7.4 (6.2)	3.4 (4.5)	4.5 (8.3)	6.2 (11.1)	1.7 (3.2)
Turbellaria	1.0 (0.9)	4.7 (3.4)	1.5 (1.7)	0.3 (0.7)	2.0 (1.3)
Caeciodotea	0.6 (0.7)	1.6 (2.2)	0.1 (0.1)	0.4 (0.8)	0.4 (0.3)
Pisidium	2.1 (1.8)	1.8 (2.1)	0.4 (0.5)	0.6 (0.5)	0.6 (0.6)
Vorticifex	0.7 (1.0)	1.1 (1.1)	0.2 (0.4)	0.5 (0.9)	0.3 (0.3)
Ferrissia	NP	NP	1.1 (3.0)	<0.1 (0.1)	1.0 (2.8)
Piscicola	0.1 (0.1)	1.0 (2.3)	NP	NP	NP
Total	**97.3**	**97.2**	**98.1**	**98.9**	**98.0**

NP, not present.

the most abundant taxa, and all are considered tolerant to organic pollution (Clark and Maret, 1993). Aquatic insects other than Chironomidae were rare in the Middle Snake River. Particularly scarce were those groups adversely affected by organic pollution, elevated water temperatures, and sedimentation, such as Ephemeroptera and Plecoptera (e.g., Clark and Maret, 1993; Merrit and Cummins, 1996). Indeed, during the course of the ISU sampling, no Plecoptera (stoneflies) were found at any Middle Snake River station. However, 8 to 10 species of Plecoptera were collected from 1981 to 1988 at various locations in Rock Creek (Maret, 1989), a tributary to the Middle Snake River.

The New Zealand mud snail, *Potamopyrgus antipodarum* (Gray) (=*P. jenkinsi* [Smith]), was first discovered in the Middle Snake River in 1987 and has spread rapidly since that time (Bowler, 1991). Sampling from 1992 to 1994 revealed *P. antipodarum* to be the most common invertebrate in the Middle Snake River. The relative abundance of *P. antipodarum* during the study at two representative stations (Rkm 986.3 and Rkm 932.4, RM 613.0 and RM 579.5) is shown in Figure 7-3. At Rkm 986.3 (RM 579.5), *P. antipodarum* represented as much as 80% of the individuals of the macroinvertebrate community. At Rkm 986.3, however, its abundance was lower, accounting for only 2% to 20% of the individuals in the community from 1992 to 1994.

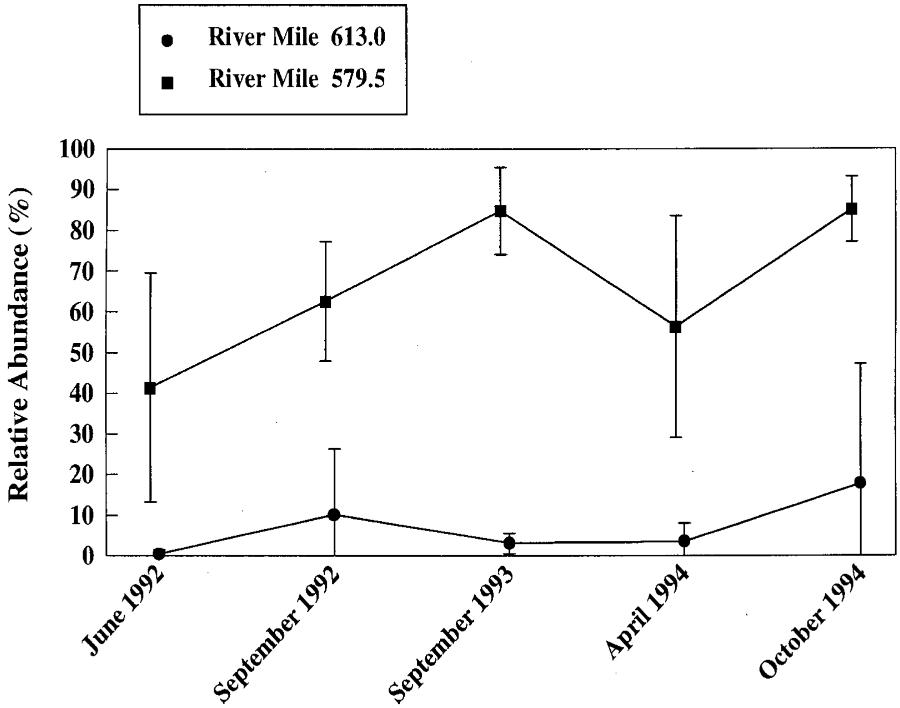

Figure 7-3. Mean relative abundance of the exotic snail, *Potamopyrgus antipodarum*, at two locations sampled by Idaho State University. Error bars equal +/- 1SD, n=9.

P. antipodarum is known to do well in eutrophic systems (Dorgelo, 1988); therefore, it is not surprising that it has been successful in the Middle Snake River.

In addition to the sampling conducted from 1992 to 1994, the macroinvertebrate community in the Middle Snake River at King Hill was sampled in August 1995 as part of a separate research project (Royer and Minshall, 1996). Sampling was limited to areas of the river approximately 1.3 m or less in depth and thus does not include deep-water habitats. At that time, the community at King Hill consisted primarily of Chironomidae, Oligochaeta, and *Hydropsyche* (a filter-feeding Trichopteran). Other benthic insects that occurred in the Middle Snake River at King Hill included *Baetis* (Ephemeroptera), *Tricorythodes* (Ephemeroptera), and *Hydroptila* (Trichoptera), all of which are moderately or very tolerant of habitat degradation. The sampling at King Hill revealed *Potamopyrgus antipodarum* to be a relatively minor component of the fauna, representing 6% of the sampled community.

7.3. STATUS OF THREATENED AND ENDANGERED MOLLUSCS IN THE MIDDLE SNAKE RIVER

Presently, cold-water natives survive only in limited, spring-fed areas in the Middle Snake River. The preferred habitat for cold-water biota has temperatures less than 17°C with minimal sediment in free-flowing water. Cold-water molluscs are most likely to be found adjacent to rapids, near spring-influenced sites, or near the mouths of major tributaries.

7.3.1. Bliss Rapids Snail (*Taylorchoncha serpenticola*)

The Bliss Rapids snail is moderately negatively phototaxic and generally resides under rocks during the day (Bowler, 1991). The Bliss Rapids snail grazes during the night on small algae and diatoms living on rock surfaces (USFWS, 1994). Egg laying takes place in December through March in the main stem and from January to late June in the large spring colonies. Eggs are laid singly in very small capsules attached to the lateral and undersides of rocks, which also are inhabited by adults. Eggs hatch roughly 1 month after oviposition. Juveniles are predominant in river colonies by early June and as early as April in large springs. There appears to be an annual turnover of most adult populations (Frest and Johannes, 1992).

T. serpenticola occurred historically in the main stem of the Snake River and associated springs from Indian Cove Bridge (Rkm 845.9, RM 525.3) to Twin Falls (Hershler et al., 1994; USFWS, 1995). The USFWS reports (1995) that colonies are concentrated in the Hagerman reach, unpolluted springs (Thousand, Box Creek, and Niagara Springs), and the tailwaters of the Bliss and Lower Salmon Falls Dams. Pentec (1991) reported finding this snail in springs above American Falls Reservoir, but Hershler et al. (1994) believe this requires verification, which has not occurred.

As of 1995, the distribution of known populations of this species was believed to be discontinuous in areas either in or influenced by springs, on the edge of rapids, and near shorelines (USFWS, 1995). In more recent work, Cazier and Myers (1996) and Cazier (1997a) observed a more or less continuous population in edgewater, runs, eddies, and deep-water habitats (5.5 to 6.7 m) from Lower Salmon Falls Dam to Bancroft Springs. However, the Bliss Rapids snail is not found in pools or reservoirs.

In aggregate, Frest and Johannes (1992) estimated the total number of Bliss Rapids snails in the Nature Conservancy's preserve in Thousand Springs to number in the low millions. At that time, it was the largest known concentration of this species. Cazier and Myers (1996) estimated relative densities of Bliss Rapids snail colonies at 3.4 to 8.2 snails/0.5 m^2 at Rkm 917 (RM 569.5) in the Snake River. In studying Bliss Rapids snail colonies during 1996, Cazier (1997a) estimated relative densities at 0.24 to 23.2 snails/0.5 m^2 in the Snake River and 49.6 to 98.6 snails/0.5 m^2 in Thousand Springs.

The Bliss Rapids snail occurs in flowing waters on exposed lateral and undersides of stable cobble-boulder substratum (Frest and Johannes, 1992). This snail does not burrow into the sediment and normally avoids surfaces with attached plants (Taylor, 1982a; Hershler et al., 1994). Taylor (1982a) found it locally quite abundant on smooth rock surfaces with encrusting algae. Cazier (1997a) found that the Bliss Rapids snail inhabits most environments available in springs and in the main river below Lower Salmon Falls Dam, but the snail is rarely found on fine sediments. Pentec (1991) reported finding 56 live specimens in 4 sites (near Rkms 941 and 1207, RM 584.4 and 749.6) with water depths of 10 to 33 cm and water currents of 6 to 100 cm/sec (facing velocity) and 43 to 122 cm/sec (0.6 depth velocity). The 56 snails were found on rock substrate 1 to 41 cm in diameter at a distance of 2.4 to 4.6 m offshore.

During 1995, dissolved oxygen ranged from 7.8 to 9.8 mg/L, temperature 7.6 °C to 19.8 °C, total hardness 172 to 195 mg/L, specific conductance 472 to 496 µmho/cm, and pH 8.3 to 8.6 in the Bliss Rapids snail colony at Rkm 917 (RM 569.5) (Cazier and Myers, 1996). These parameters were also measured at this location during 1996 (Cazier, 1997a). Dissolved oxygen ranged from 7.8 mg/L in July to 11.2 mg/L in February, temperature ranged from 4.6°C in December to 19.3°C in July, total hardness 140 to 212 mg $CaCO_3$/L, specific conductance 354 to 514 µmho/cm, and pH 8.2 to 8.8. Pentec (1991) measured dissolved oxygen at 6.9 to 8.6 mg/L in the four upriver snail beds they reported finding. Water quality conditions required to maintain reproduction and recruitment for the Bliss Rapids snail currently are unknown.

7.3.2. Idaho Springsnail (*Pyrgulopsis idahoensis*)

Information is not available on the life history of this species. The historical distribution of the Idaho springsnail was in permanently flowing waters of the mainstem Snake River from Homedale (Rkm 670, RM 416) to Bancroft Springs. It was not found in tributaries to the Middle Snake River or in marginal cold-water springs (Taylor, 1982b; USFWS, 1995). Taylor (1982b) collected this species from Bancroft Springs to Indian Cove Bridge, a distance of about 45 km. The consistent occurrence of this snail in Taylor's samples led him to believe its distribution was nearly continuous along this reach of the Snake River.

As of 1995, Idaho springsnail populations were believed to occur in approximately 20% of their historical range and were discontinuous in the reach from the headwaters of C.J. Strike Reservoir (Rkm 834, RM 517.9) to Bancroft Springs (USFWS, 1995). More recent investigations by Cazier and Myers (1996) and Cazier (1997a,b) have found the Idaho springsnail in a 34-km reach below C.J. Strike Dam, within the C.J. Strike Reservoir, and in the 5-km reach above Bancroft Springs. These investigations expanded the range of the Idaho springsnail, which now appears to extend from Rkm 761 (RM 472.6) to Rkm 895 (RM 555.8).

Taylor (1982b) believed the upriver limit for the Idaho springsnail populations was the fast water in a narrow canyon section of the Snake River above Bancroft Springs and the downstream limiting factors were unknown. Cazier and Myers (1996) estimated relative densities of Idaho springsnail at 0.63 to 1.5 snails/0.5 m^2 at Rkm 895 (RM 555.8) in the main river. Cazier (1997a) estimated densities from 10 to 60 snails/0.5 m^2 in Idaho springsnail colonies in the C.J. Strike Reservoir and in the 34-km reach below the dam.

The Idaho springsnail is found in mud or sand among cobbles and boulders and its probable habitat includes the entire width of the Snake River, except in strictly coarse-grained sediments (Taylor, 1982b). After observing this snail living at water depths of 15 cm to 7 m on cobble, gravel with or without vegetation, on mud or sand between cobble, and on gravel covered with algae, Cazier (1997a) believes this snail is an ecological opportunist. Further, Cazier (1997b) found *P. idahoensis* occurring in the fluctuation zone (area <2 m deep affected by block loading) in the Snake River below C.J. Strike Dam.

During 1995, dissolved oxygen ranged from 7.8 to 9.8 mg/L, temperature 7.6 °C to 19.8 °C, total hardness 172 to 195 mg/L, specific conductance 472 to 496 µmho/cm, and pH 8.3 to 8.6 in an Idaho springsnail colony in the main river (Cazier and Myers, 1996). Cazier (1997a) reported finding Idaho springsnail colonies in C. J. Strike Reservoir in an area with summer water temperatures of more than 22°C. Water quality conditions required to maintain reproduction and recruitment for the Idaho springsnail are currently unknown.

7.3.3. Snake River Physa (*Physa natricina*)

Although little is known of the general life history of the Snake River Physa, longevity probably averages 2 years (USFWS, 1994). The distribution of this species is based on the collection of only a few live specimens and empty shells. The historical distribution of the Snake River Physa was believed to extend from Grandview (Rkm 783.4, RM 486.5) upstream through the Hagerman Reach (Rkm 917, RM 569.5) (USFWS, 1995). The range of this snail may extend further upriver than reported by the USFWS as Pentec (1991) reported finding two live snails at two new sites (Rkm 1191 and 1205, RM 739.6 and RM 748). This appears to be a very rare snail with less than fifty specimens collected. Population densities are not available. At present, two colonies are believed to remain in the Hagerman and King Hill reaches, and a possible third colony may be located immediately downstream of Minidoka Dam (USFWS, 1995).

The Snake River Physa lives primarily in deep, swift water. Taylor (1982c) found live snails on boulders in the deepest accessible part of the Snake River at the margins of rapids. Pentec (1991) reported finding two snails on substrate 0.7 and 5 cm in diameter at locations 7 and 30 m offshore during low-water conditions. The water at these locations was 5 and 46 cm in depth, and the water currents were 30 and 46 cm/sec (facing velocity) and 46 and 52 cm/sec (0.6

depth velocity). The Snake River Physa requires cold, clean, well-oxygenated, swiftly flowing water with low turbidity (USFWS, 1995). Pentec (1991) measured dissolved oxygen at 7.7 and 8 mg/L at the two locations where Snake River Physa were found. Water quality conditions required to maintain reproduction and recruitment for this species are currently unknown.

7.3.4. Utah Valvata (*Valvata utahensis*)

V. utahensis is primarily a detritivore, grazing on diatoms and other periphyton (USFWS, 1995). Little is known of its life history, but it is believed to have a maximum longevity of 2 years (USFWS, 1994). The "modern" historical range of Utah valvata extended from Grandview upstream to American Falls (USFWS, 1995). The present distribution of this snail includes springs and the main-stem Snake River in about a 6-km reach of the Hagerman Valley, a 72-km reach of Lake Walcott, and a site below American Falls Dam (Taylor, 1982d; USFWS 1995; Bureau of Reclamation, 1998).

The Bureau of Reclamation (1998) found that Lake Walcott serves as a refuge for a large Utah valvata population. In 1996 surveys, Ralston and Associates (1997) reported average population densities of 0.5 to 4 snails/m^2 (range 0.09 to 15.1 snails/m^2) for live Utah valvata colonies in Lake Walcott. The highest density was near Eagle Rock and the lowest density was recorded between Jackson Bridge and Minidoka Dam. Overall, the Lake Walcott reservoir average density was 1.8 snails/m^2, whereas density averaged 0.5 snails/m^2 in the reservoir/spring sites.

Surveys by Frest and Johannes (1992) in the Nature Conservancy's Thousand Springs Preserve found average population densities of about 0.25 snails/m^2 in two colonies with totals of about 6,000 snails per colony. Frest and Johannes (1992) believe these colonies were in decline. Taylor (1985) concluded that sediments deposited in a natural pool in lower Box Canyon from the construction of an earthen diversion dam upstream in 1973 increased the habitat for Utah valvata but severely degraded the habitat for the Bliss Rapids snail.

According to Taylor (1982d), this species inhabits large streams and lakes, where it occurs on mud, fine sand, and silt bottoms. Taylor (1982d) found Utah valvata living on fine silt among beds of submerged aquatic plants (*Potamogeton*, *Ceratophyllum*, *Myriophyllum*, and sedges) in the Hagerman Valley. USFWS (1995) reports Utah valvata inhabits pools adjacent to rapids in the main river or in perennial flowing waters associated with large spring complexes, but avoids areas of high water velocity and rapids. The snail prefers well-oxygenated areas of limestone mud or mud-sand substrate among beds of submerged vegetation (USFWS, 1995). Cazier (1997a) found that this snail burrows into mud/sand substrate and suggested the snail is a generalist, not a specialist, in terms of habitat requirements. In Lake Walcott during 1997, the USGS found Utah valvata in 80 of 195 (41%) sites; the snails were present in water depths ranging from 0.8 to 13.8 m. Seventy-five percent (60 of 80) of the sites were at depths greater than about 3 m (Bureau of Reclamation, 1998).

During 1996-97 field sampling in the Snake River just below Minidoka Dam and in Lake Walcott, the Bureau of Reclamation (1998) measured six water quality parameters at sites with Utah valvata. Their observations found this snail living in Lake Walcott under the following ranges of water quality conditions: water temperatures from 4.5°C (in November) to 18.2°C (in June); dissolved oxygen from 8.3 mg/L (in June) to 16.1 mg/L (in November); pH from 7.5 (in November) to 9.0 (in October); specific conductance 334 µmho/cm (in June) to 897 µmho/cm (in November); total alkalinity 123 mg $CaCO_3$/L (in June) to 155 mg $CaCO_3$/L (in November); and total hardness 129 mg $CaCO_3$/L (in June) to 345 mg $CaCO_3$/L (in November). The colony of snails living in the river below Minidoka Dam was found in similar water quality conditions, except they were exposed to higher temperatures (19.2°C in August) and lower dissolved oxygen (6.8 mg/L in August). Water quality conditions required to maintain reproduction and recruitment for the Utah valvata currently are unknown. However, it appears that this snail is capable of living in lake or reservoir conditions and does not require cooler, riverine conditions.

7.3.5. Banbury Springs Lanx (*Lanx sp.*)

This mollusc, as with other lancids, appears to feed exclusively on aufwuchs. It occurs primarily on the lateral and undersides of rocks, but not in contact with the sediment (Frest and Johannes, 1992). For most of the population, a 1-year life span is typical and adults experience mortality during the late winter or early spring following reproduction (Frest and Johannes, 1992; USFWS, 1994). As with other lancids, oviposition likely occurs 1 month after copulation, which has been observed only in the spring. Eggs of *Lanx* sp. are laid only on rocks and have been seen from April through June. The egg capsules are small (no more than 1.5 mm in diameter) with no more than 6 eggs per capsule. Juveniles have been observed from May through July and growth is most rapid between July and October.

The Bliss Rapids snail commonly occurs in association with the Banbury Springs lanx (Frest and Johannes, 1992). After being first discovered in Banbury Springs at Rkm 949 (RM 589) in 1988, this lanx (or limpet) was found in two other alcove spring complexes: Box Canyon Springs near Rkm 946 (RM 588) and in Thousand Springs near Rkm 941 (RM 584.4) (Pentec 1991; USFWS, 1995). With one possible exception, this species is known to occur only in these large, undisturbed springs. With regard to the possible exception, Beak (1987) reported finding 20 individuals of *Lanx* sp. in the main stem of the Snake River at Rkm 889.5 (RM 552.3). Subsequent references on the distribution of the Banbury Springs lanx, however, have not recognized this finding.

In a June-July 1991 survey in the Nature Conservancy's Thousand Spring Preserve, Frest and Johannes (1992) found a small Banbury Springs lanx colony in an area covering roughly 14 m^2. The limpets were very sporadically distributed in the area; average densities ranged from 16 to 48 individuals/m^2. The total number of adults in the colony was estimated to be between 600 and 1,200 individuals. Descriptions of the colonies in Box Canyon and Thousand Springs are not available.

The Banbury Springs lanx is found only in spring-run habitats and avoids areas with large, attached plants or areas with fluctuating water levels. This species is typically found in water depths of 15 cm, but it did occur in water as shallow as 5 cm deep. All sites had relatively swift water currents with mostly smooth basalt boulders having a maximum dimension of at least 7 cm. The Banbury Springs lanx avoids granites and rocks with vesicular textures (all from Frest and Johannes, 1992). During March 1991 in Thousand Springs, Pentec (1991) reported finding five live individuals of *Lanx* sp. on rock substrate estimated to be 25 to 40 cm in diameter at a location 6 m offshore. This site had a water depth of 40 cm and water currents of 33.5 cm/sec (facing velocity) and 51.8 cm/sec (0.6 depth velocity).

This lanx is a member of the family Lancidae, a small group of pulmonates that respire solely through a highly vascularized mantle. As specialized respiratory organs are lacking, the lancids are particularly susceptible to fluctuations in dissolved oxygen (Baker, 1925; from Frest and Johannes, 1992). The three sites in Banbury Springs where Frest and Johannes (1992) collected this species had well-oxygenated, clear, cold (15 °C to 16°C) water. However, Pentec (1991) measured dissolved oxygen at 6.9 mg/L at the site surveyed in the Thousand Springs.

8. ANALYSIS OF EXPOSURE AND EFFECTS FOR AQUATIC PLANTS

8.1. HISTORIC TRENDS

Early works addressing biotic conditions in the Middle Snake River generally ignored aquatic macrophytes, focusing instead on game fish, chemical water quality, and eventually, attached benthic algae and benthic macroinvertebrates (Gebhards, 1969; U.S. EPA, 1974; Falter et al., 1976). These reports made no, or only passing, mention of aquatic macrophytes in this reach. Known high macrophyte density areas in the mid-1970s were in the shallow upper reaches of Milner Pool, Box Canyon, and the upstream reach of Upper Salmon Falls Reservoir. The 8-year extreme low-flow period of 1986-93 undoubtedly exacerbated the long-term trend of riverbed shallowing and subsequent development of high density macrophyte beds where they had been little noticed in earlier studies. Idaho Power (1995) assessed macrophytes in the three reservoir reaches from 1987 aerial photography and found that approximately 20% of Upper Salmon Falls Reservoir was covered with submerged or floating plant beds. Macrophyte densities declined downstream in Lower Salmon Falls and Bliss Pools. Idaho Power described 15 species of macrophytes (including all of the species found by Falter and Carlson, 1994; Falter et al., 1995; and Falter and Burris, 1996), but did not address epiphytes.

The Federal Energy Regulatory Commission (1990) stated that "...overall, however, plant production in the study reach (Milner-Star Falls) is very high, especially during the low-flow, summer irrigation season, when warm water, high nutrients, low flow, and high light provide ideal conditions for the plants." The Federal Energy Regulatory Commission (1990) commented that submerged macrophytes occupied 23.4% of Lower Salmon Falls Reservoir, but only 2.1% of the more turbid Wiley reach. Idaho Power (1990) noted that shallow, low-velocity areas of Bliss, Lower Salmon Falls, and Upper Salmon Falls Reservoirs supported dense macrophyte populations, but related their occurrence more to areas "...where spring water enters the main stem." Idaho Power thought that "aquatic macrophytes in Upper Salmon Falls and Lower Salmon Falls reservoirs are more prevalent than would be expected in a free-flowing system, and the aquatic plants may affect water quality and recreational value of the reservoirs." Hill (1991c) delineated macrophyte beds in the Middle Snake River by using aerial photographs and GIS maps. He found 20% aerial coverage of the river between Crystal Springs and Banbury Springs. *Ceratophyllum* and *Potamogeton* spp. were the dominant taxa. Falter and Carlson (1994), Falter et al. (1995), and Falter and Burris (1996) described composition, biomass, and sediment characteristics of the reach from on-ground studies.

8.1.1. Phytoplankton

Summer chlorophyll *a* concentrations in the main river are in the mesotrophic to eutrophic range, mostly 2 to 20 mg/m^3 (2 to 10 mg/m^3 for mesotrophy and >10 mg/m^3 for eutrophy as defined for lakes by U.S. EPA, 1990). Using chlorophyll *a* level for lakes is reasonable in the Middle Snake because the decreased flows cause the river ecosystem to resemble lentic dynamics with nutrients being readily available. Mean chlorophyll *a* concentrations in the Crystal Springs reach (3.7 to 12.1 mg/m^3) showed little variation with the exception of the above-the-plant bed site in July. Concentrations were comparable to those in the Box Canyon reach (1.8 to 19.3 mg/m^3). Mean chlorophyll *a* concentrations in the Box Canyon reach were in the mesotrophic range.

Model sensitivity analysis showed the dynamics of phytoplankton in the Middle Snake River to be more responsive to hydrology and to initial conditions from upstream sources. There was little or no correlation between chlorophyll *a* concentrations and physical or chemical water quality variables in the Middle Snake River, with the exception of total Kjehldahl nitrogen (TKN). Regression analysis showed a positive relationship between TKN concentrations in the water column and chlorophyll *a* in both the Crystal Springs and Box Canyon reaches.

The mean chlorophyll *a* levels in the river transects below the springs were markedly lower than in the main river. Chlorophyll *a* concentrations ranged from 0.5 to 1.6 mg/m^3 for all below-springs transects. The lowest concentrations were measured along the Box Canyon Spring transect in June and August (0.5 mg/m^3 for both months). Banbury Springs and the Nature Conservancy Springs' transects both showed a decrease in mean chlorophyll *a* concentrations from June to August. Concentrations along the Banbury Spring transect dropped from 1.5 to 1.3 mg/m^3, and the concentrations along the Nature Conservancy Spring transect dropped from 1.3 to 0.7 mg/m^3 from June to August.

8.1.2. Vascular Macrophytes

The most detailed treatment of aquatic macrophytes in the Middle Snake River has been provided by the studies of Falter and Carlson (1994), Falter et al. (1995), and Falter and Burris (1996). These studies, however, only addressed the reach from Twin Falls downstream to Upper Salmon Falls Dam, with most coverage from Crystal Springs reach and Box Canyon reach. The following information is taken from those studies. These described reaches are believed to represent the most dense occurrences, but composition and habitat descriptions as well as macrophyte controlling factors are thought to be representative of the entire Middle Snake reach. Plant material at all sites during summer in the Crystal Springs reach was dominated by epiphyton (Figure 8-1). Epiphytes made up <1% of the plant community (Table 8-1) early in the

Figure 8-1. Snake River at Crystal Springs, July 1992.

Table 8-1. Riverwide mean plant biomass and percent of the total biomass, Crystal Springs, Idaho, 1994 (from Falter and Burris, 1996)

	Epiphytes, mean g/m^2	Non-rooted, mean g/m^2	Rooted, mean g/m^2	Total, g/m^2	Epiphytes, % of total	Nonrooted, % of total	Rooted, % of total	All types, total
Apr	0.1	34.0	30.6	64.7	0.2%	52.5%	47.3%	100.0%
May	1.3	3.6	230.8	235.7	0.6%	1.5%	97.9%	100.0%
Jun	231.5	78.5	239.2	549.2	42.2%	14.3%	43.5%	100.0%
Jul	163.4	25.6	137.1	326.1	50.2%	7.8%	42.0%	100.0%
Aug	173.8	248.9	81.8	504.4	34.5%	49.3%	16.2%	100.0%
Oct	214.4	138.3	104.1	456.7	46.9%	30.3%	22.8%	100.0%
Nov	63.3	46.6	52.9	162.8	38.9%	28.6%	32.5%	100.0%

season, but shifted to approximately 50% dominance by midsummer to late fall. Rooted macrophytes dominated the plant community early in the year, accounting for up to 98% of total composition in May, before gradually declining as a percent of total composition later in the year. In August, rooted macrophytes were only 16% of total macrophyte biomass. The epiphyton component gradually decreased at all sites through the summer.

In the Box Canyon reach, *Potamogeton crispus* was dominant at above-plant bed and open-channel sites, ranging from 19% to 47% of total biomass. Epiphyton was virtually absent from both the open-channel and the above-plant bed sites, but was a dominant component in the plant bed sites. The exceptions were in the open-channel site in August when *P. foliosus* was dominant and at the above-plant bed site in July when no plants were collected.

In the Thousand Springs reach, macrophyte diversity was higher than in Crystal Springs and not as dominated by epiphyton. The dominant macrophyte in the littoral plant bed area was *E. nuttallii* in July (20%) and *C. demersum* in August (32%).

Mean percent composition of the aquatic plant community in the springs and in localized areas of the river channel where water quality is dominated by springs was strikingly different from that in the main river. Composition of aquatic plant communities in the alcove springs along the Middle Snake River is more diverse and balanced than the communities in the main channel of the Middle Snake River. Epiphyton was still a component of total biomass, albeit a minor component and not nearly as dominating as in the river, especially in upstream reaches. There was a much wider variety and evenly distributed array of species represented in the springs. *Myriophyllum spicatum* var. *exalbescens* was present in three of the springs and was found nowhere else in the main river. The moss *Drepanodacladius sp.* (requiring clear, cold water rich in CO_2) and *Ranunculus* sp. along with *M. spicatum* var. *exalbescens* all were found only in the unique environment of the springs.

Box Canyon Spring had the most diverse macrophyte composition. There were five species making up 8% or more of the aquatic plant biomass present in June. The plant community in the Nature Conservancy Spring also consisted of a broader range of species compared with the main river reaches.

8.2. DENSITIES OF PLANT COMMUNITIES IN THE MIDDLE SNAKE RIVER

In the early 1990s, plant beds in the Middle Snake River would develop very high biomass levels (up to 3,000 g/m^2), greater than those considered "nuisance levels" on the Pend Oreille River, WA (Coots and Willms, 1991; Falter et al., 1991). The results of our survey (Table 5-2) suggest that 200 g/m^2 average maximum biomass as ash-free dry matter (AFDM) is a reasonable lower bound for nuisance levels of aquatic macrophytes.

Heavy plant beds in the Middle Snake River are found downstream of agricultural return flows and/or aquaculture discharges, with the heaviest growths downstream of the combination of both discharges. In the immediate vicinity of irrigation drain discharges to the river, sediments are predominantly fine sands and silts. High sediment deposition rates and increased turbidity in the first several hundred yards below discharge points preclude aquatic macrophyte development. After a space of low growth, plant beds increase in size and density. Similarly, the first 90 to 200 meters downstream of aquaculture effluent discharge points are a sediment deposit area of fine sands and silts, but with a higher organic content. This immediate area is generally devoid of plant growth except for about a 0.2-m-thick bed of *Cladophora* sp. growing on the sediments. In these latter instances, the anaerobic, reducing nature of the organic-rich sediments apparently impedes rooted plant growth even though sediment nutrient concentrations are high. Further downstream, about 90 to 200 meters, very high plant aquatic plant densities begin to occur. Unless indicated otherwise, the following mean biomass values from summer to fall 1992 are derived from three to eight replicate samples per area or transect.

Summary plant biomass values in the Crystal Springs reach show the following summer-long downstream trend of site means:

Above-plant bed (upstream end of reach)	-	348 g/m^2 *
Left channel (mid-reach, in plant beds)	-	784 g/m^2
Right channel (mid-reach, in plant beds)	-	772 g/m^2
Top-of-plant bed (downstream end of reach)	-	1,551 g/m^2
Bottom-of-plant bed (downstream end of reach)	-	1,141 g/m^2

* Only littoral (<2.0 m depth) samples were included in all means in this data set.

Summer mean plant biomass increased fourfold downstream through the 2.9-km (1.8-mile) Crystal Springs reach. The highest mid- and downstream densities were below a series of aquaculture effluent discharges. No other effluents came into this (2.9-km) 1.8-mile reach during the sampling years. Means were also significantly different between the top and the bottom of the plant bed.

Table 8-1 summarizes 1994 aquatic macrophyte mean dry biomass over all transects and months (Falter and Burris, 1996) for the Crystal Springs reach. The dry weights in g/m^2 are overall river means, separated into three functional groupings: epiphytic algae, nonrooted vascular macrophytes, and rooted vascular macrophytes. Epiphytic algae and nonrooted vascular macrophytes may be considered functionally grouped, as they absorb their nutrients from the water column rather than from sediments. The following overall river means are true average concentrations over the entire river, plant-covered and plant-free areas alike, and are more appropriate for use in a river ecological model that considers the entire river bottom.

Biomass of all plant types combined (i.e., mean total aquatic macrophyte biomass) in Crystal Springs reach increased from 65 g/m^2 in April to an average of 459 g/m^2 in the summer-fall months of June, July, August, and October. Biomass fell to 163 g/m^2 in November. High-biomass months were June and August with 549 and 504 g/m^2 mean biomass, respectively. Epiphytes followed a similar trend, increasing from <1 g/m^2 in April to a summer-fall mean of 196 g/m^2 (June to October). High epiphyte months were June and October, with 232 and 214 g/m^2 mean biomass, respectively. Nonrooted vascular macrophytes developed later in the season, averaging only 35 g/m^2 April through July to an annual peak mean of 249 g/m^2 in August. True rooted macrophytes developed first, attaining an average of 231 g/m^2 by May before steadily declining to 53 g/m^2 in November.

Mean aquatic plant biomass in the Box Canyon reach, a slightly deeper site than Crystal Springs, followed a different pattern. Mean biomass in the Box Canyon reach above-plant beds site was near zero for June and July. Mean biomass at the above-plant beds site increased to 14 g/m^2 in August, a time when water transparency had increased to 2.0 m. Mean aquatic plant biomass in the open channel decreased gradually through the summer from 31 g/m^2 to 20 g/m^2. Plant biomass in these two sites would be expected to be less than in Crystal Springs because of greater mean depths in Box Canyon. Mean aquatic macrophyte biomass in the plant bed in June was about 225 g/m^2 and increased to 337 g/m^2 in July. Aquatic plant biomass in the plant bed decreased sharply in August to a mean of 68 g/m^2.

Aquatic plant biomass in the shallower Thousand Springs reach was higher than in Box Canyon. Mean biomass in the littoral plant bed area in July was 314 g/m^2, increasing to 483 g/m^2 in August.

Mean aquatic plant biomass in the river transects at the springs showed wide cross-channel biomass variation as water quality gradients changed in very short distances across the river. Lowest biomass measurements were from Blue Heart Spring transect for both June and August (13 and 102 g/m^2 ,respectively). Highest biomass measurements from spring transects were in June (345 g/m^2) and August (876 g/m^2). The remaining two transects, Banbury and Nature Conservancy Springs (immediately upstream, and contiguous to Box Canyon and Thousand Springs reaches, respectively), had August biomass means (612 and 478 g/m^2, respectively) that were as high or higher than those in the Box Canyon and Thousand Springs reaches (337 and 483 g/m^2, respectively).

Blue Heart Spring (Figure 8-2) transect had mean plant biomass values of 56 g/m^2 in June and 97 g/m^2 in August. Box Canyon Spring transect had plant biomass values of 1,376 g/m^2 in June and 961 g/m^2 in August. Banbury Spring transect had plant biomass values of 79 g/m^2 in June and 1,885 g/m^2 in August. The Nature Conservancy Springs transect had plant biomass values of 776 g/m^2 in June and 777 g/m^2 in August.

Figure 8-2. Blue Heart Springs with Box Canyon in the background, Middle Snake River, August 1993.

Mean plant densities in the mid-1990s studies were generally lower in springs than in the river, even with the very clear water (Falter and Carlson, 1994; Falter et al., 1995; and Falter and Burris, 1996). Cross-river transects at the springs showed that Blue Heart, Box Canyon, Banbury, and Nature Conservancy Springs in the Middle Snake River were all high water quality, cold-water habitats of significantly lower mean water column and sediment nutrient concentrations than the main channel of the Middle Snake River.

8.3. FACTORS CONTROLLING PLANT GROWTH, BIOMASS, AND DIVERSITY

There are many environmental variables that both shape plant communities in the Middle Snake River and are shaped by the plant communities.

Correlation of total aquatic macrophyte biomass with the physical variables of depth, temperature, and velocity did not show significant relationships. By including *Ceratophyllum demersum* with the epiphyte component, a strong relationship was identified between the epiphytic portion of total aquatic macrophyte biomass and water velocity. This grouping was based on the fact that *Ceratophyllum demersum* does not have a well-developed root structure and obtains its nutrients in a fashion similar to filamentous epiphytic algae, i.e., from the water column (Kennedy and Gunkel, 1987).

Plant biomass values were significantly related to soluble reactive phosphorus (SRP) only in the Crystal Springs reach. The correlation was observed in both linear and second degree polynomial models (Falter and Burris, 1996). The second-degree polynomial demonstrated a significant correlation (negative) between plant biomass and SRP ($r^2 = 0.51$, $F = 31.39$, $n = 64$) in the Crystal Springs reach.

In the Middle Snake River, as in most aquatic macrophyte situations, there is generally poor correlation of dissolved nutrients to aquatic plant biomass. Notable exceptions are soluble reactive phosphorus in the Crystal Springs reach and nitrite/nitrate in the Thousand Springs reach. The reason for this poor correlation is the inherent complexity of aquatic systems. A rooted macrophyte may well draw the majority of its nutrient needs from the sediments most of the time, but when other aspects of the system are also considered, it is apparent that high-water-column nutrients eventually result in higher macrophyte coverage, both in terms of growth rates and aerial coverage. There are several major considerations:

- Rooted macrophytes do not exist in a vacuum, but in a complex community of many plants with different nutritive strategies.
- Epiphyte cover on the rooted plant (both unicellular and filamentous) absorbs nutrients from the water column and builds sediments upon the plant's senescence.
- Epiphytes can pass nutrients to foliar surfaces of their supporting plants.
- Nonrooted vascular macrophytes absorb nutrients from the water column, build biomass, and contribute to building nutrient-rich sediments supportive of rooted macrophytes.
- A matrix of aquatic macrophytes, whatever mix of rooted, nonrooted, or epiphytes, filters planktonic algae and nutrient-rich detritus from the water column, building sediments for further nutrition of rooted macrophytes.

True rooted macrophytes developed earlier in the year (May) than nonrooted and epiphytic macrophytes in the Middle Snake River. It is likely that the nutrient and energy reserves of the perennial root masses permit earlier, faster plant development than in epiphytes and rootless plants. Epiphytic algae cannot reach high biomass levels until June, when epiphytes can obtain three required conditions: (1) high enough temperatures (Millner et al., 1982), (2) sufficient nutrients from the water column, and (3) adequate physical support provided by rooted plants high enough in the water column to obtain their required high light nearer the water surface. By midsummer, the combined community of epiphytes and nonrooted macrophytes apparently outcompetes rooted forms. The faster rates of nutrient uptake by the entire nonrooted plant and the light-favored position higher in the water column permit epiphytes to avoid light limitation and ensure their success. Considering their absence of nutrient-absorbing roots, the epiphytes and nonrooted forms, which are dominant in late summer and fall, can only attain high biomass levels with high-water-column nutrients. Howard-Williams (1981) found that fertilization with N and P sharply increased the development of epiphytes (*Cladophora*) on *P. pectinatus* communities, but did not increase the rooted plant biomass.

Low nutrient thresholds of sediment TKN, sediment phosphorus, and water column SRP probably dictate low nutrient limitation for controlling plant biomass; *high* nutrient thresholds of sediment TKN are related to other direct controllers of plant biomass, i.e., anaerobic sediments, high sediment levels of ammonia, hydrogen sulfide, and carbon dioxide (Falter and Carlson 1994, Falter and Burris 1996). This would explain the near absence of most macrophytes (except for *Cladophora* spp) in areas of the Middle Snake River accumulating organic sediments from aquaculture facilities. Barko (1983) similarly found that high-level accumulations of organic matter in sediments inhibited milfoil growth. He concluded that the low redox potentials, accumulations of organic acids, low pH, increased metal availability, and evolution of growth-inhibiting gases in highly organic sediments overshadowed the high fertility provided by a high-organic-matter and high-nutrient environment.

The distribution of vascular macrophytes in the Middle Snake River is controlled primarily by available light, which in turn is controlled by water clarity and depth, sediment particle size, sediment nutrient concentrations, and water velocity. Also, the plant beds themselves have a major feedback role in creating desirable water velocity and sediment conditions. Water column nutrient concentrations contribute to this process via the mechanisms described above. Plant beds occur in the Middle Snake River only where water depth is $\leq$ 6 m. Heaviest plant densities were in the 0.5 to 2.0 m depth range over flat to very gently sloping bottom. Water velocities in the 0 to 1.0 meters per second (mps) range were ideal for plant development. At deeper depths, light availability is apparently the limiting factor to occurrence of significant plant beds. Where depth is not limiting, water transparency $\geq$ 0.7 m is required for significant development of plant beds. Moderate to high turbidity levels in the main river often limit high plant production in the channel. In waters of $\leq$ 6 m, ideal sediment conditions for plant development are where sediments are in the fine sand-silt-clay range of particle size.

Given the above conditions, true rooted vascular macrophytes may start as an island of rooted stems, even in high water velocity as with *Potamogeton pectinatus*. As the plant bed develops in size and stem density, intrabed conditions are increasingly shaped by the plant community as described above. In the protected "nursery" environment within a bed, low-velocity-preferring forms such as *Elodea* and *Ceratophyllum* (nonrooted plants) can then develop. Floating epiphytes seem to do best at intermediate water velocities in the Middle Snake River. In the heavier plant beds, water velocity drops to near zero compared with 0.5 to 1.0 mps in open channels only several decimeters away. Near-bottom velocities are especially low, permitting further settling of fine particles.

In 1993 and 1994, Falter et al. (1995) and Falter and Burris (1996) found sediment total phosphorus in the Middle Snake River to be strongly related to plant biomass. Maximal plant biomass occurred at a total phosphorus level of approximately 1,100 mg/g dry weight. This trend was seen throughout areas in the Middle Snake River where sediment composition was studied. With the higher sediment phosphorus levels in downstream transects (up to and beyond 1,000 mg/g), plant biomass increased to peak levels observed. Biomass tended to decline at sediment phosphorus >1,100 mg/g. There were well-defined upper limits of sediment organic matter (4.0%) and sediment nitrogen (0.35%) above which macrophyte densities declined. High concentrations of nutrients are not likely to directly cause plant biomass decline. The direct limiting factors are likely other parameters associated with extremely high sediment nutrients and organic matter, i.e., low sediment oxygen, redox potential, and pH; and higher ammonia, hydrogen sulfide, hydrogen (i.e., reduced organic matter), organic acids, and carbon dioxide levels.

8.4. EFFECTS OF EXCESS GROWTH ON THE MIDDLE SNAKE SYSTEM INCLUDING EUTROPHICATION

Diel oxygen levels may fall below 6.0 mg/L with pH sometimes >9.0 in heavy plant beds, but oxygen stress has not been documented in these dense Middle Snake plant communities. Plant densities may be high enough so that significant nutrient depletion may be seen from upstream to downstream within some of the heavier plant beds, such as Crystal Springs and Box Canyon (Falter and Carlson, 1994; Falter and Burris, 1996).

In the Crystal Springs reach of the Middle Snake River, total phosphorus (TP) declined at a rate of 0.22 mg/L/kilometer from upstream to downstream through a dense, mixed-species plant bed in August, which was the month of maximum standing crop (Falter et al., 1995). At that time, SRP declined at a rate of 0.04 mg/L/km through the same plant bed. SRP increased downstream through macrophyte beds in response to reach loading in June and November, months of low plant density, but sharply decreased downstream through plant beds in July, August, and October, months of high density. These observations show that Middle Snake River plant beds were clearly removing TP and SRP from the water column at times of high plant densities. The nutrients do not leave the river, but are stored in the plant material. During senescence, leaching nutrients contribute to the cycle of excessive growth.

Mean aquatic plant biomass (oven dry weight) was highest in the Crystal Springs reach, with mean values for some sites in excess of 2,000 g/m^2 (Falter and Carlson, 1994; Falter et al., 1995). No sites in any of the other reaches approached this level. The next highest biomass measurements were from Box Canyon Springs proper. The distribution of macrophytes within the reaches was uneven and limited to waters ≤ 4.0 m deep. There were some exceptions with macrophytes growing in waters > 4.0 m depth. These exceptions included the Box Canyon and Thousand Springs sites where sparse growths of *Potamogeton crispus* occurred in waters up to 6.0 m deep.

Mean sediment TKN and TP were highest in the Crystal Springs reach. Aquatic plant biomass was up to eight times greater in the Crystal Springs reach than in any other site. Very high sediment nutrient concentrations (TKN and TP) are also found in the Snake River proper at the head of Upper Salmon Falls Pool (Falter and Carlson, 1994; Falter et al., 1995). Sediment nutrient and TOC (total organic carbon) levels in the pools were lower than in the adjacent shallower reaches. This reflected sediment trapping by macrophytes in the upstream shallow plant bed reaches.

The combination of deep, fine, and nutrient-rich sediments downstream of those areas in the Middle Snake River receiving organic and nutrient loading favors macrophytic plant growth. Once beds develop in these regions, internal water velocities are slowed, resulting in further sediment deposition. The end result is that sediments beneath plant beds in the Middle Snake River are fine (<6 mm), nutrient-rich, organic-rich, and oxygen-stressed. The percentage of very fine (<75 µm) sediment particles in the plant beds is even greater than in the deeper pool areas. Probing of sediments beneath plant beds in the Crystal Springs found up to 3 m of fine sediments overlying a cobble stream bed (Falter and Carlson, 1994; Falter and Burris, 1996).

Initial deposition of sediments is key to the high levels of macrophytes observed in the early- and mid-1990s. Deposits of fine, largely inorganic sediments, mostly from irrigation return flows, cause shallowing of extensive reaches until water depths approach approximately 2 m. At that point, sufficient light reaches the bottom to support macrophyte growth. These macrophyte growths, as dense plant beds, trap sediments and detritus from the water column, enriching sediments with organic matter and nutrients. By midsummer, very thick epiphytic algae mats develop on the vascular macrophytes, taking advantage of water column nutrients in the enriched river to further build plant biomass. This epiphyte biomass is later incorporated into sediments, supplementing nutrient levels at a time when rooted plant uptake has started to deplete sediment nutrients from the root zone. Through this late-summer high-growth period, however, sediment nitrogen and phosphorus concentrations decline slightly, apparently a result of plant uptake. During this period, in the bottom areas devoid of plants, nitrogen and phosphorus concentrations continue to build through the fall (Falter and Burris, 1996). The fall senescence of some plants (such as the early-fading *P. crispus*) enhances the environment for continued plant bed succession.

An important result of these plant beds in the Middle Snake River is the overall slowing of organic carbon turnover rates. Thomas et al. (1995) found that despite very high rates of respiration in the Middle Snake River, which are 1 to 2 orders of magnitude higher than the Kootenai River, the river was turning over its organic carbon pool very slowly, in fact at about 10% of the rates measured in the Kootenai River. This indicates that intense carbon utilization occurred, as might be expected in a system receiving allochthonous loadings. Turnover time for carbon is the time required for the average carbon molecule to pass from being fixed in photosynthesis to being released via oxidative decomposition. This indicates intense trapping of detritus, energy, and nutrients in the Middle Snake River, a process facilitated by the sediment deposits and abundant, dense plant beds. The classic downstream nutrient spiral from water column into biota and back to the water column is thereby shortened in the Middle Snake River, with intense cycling within plant beds and less flow of nutrients or organic carbon downstream.

9. RISK CHARACTERIZATION

Risk characterization is the final phase of the ecological risk assessment. At this point, the lines of evidence and likelihood of recovery for each assessment endpoint are summarized. The sources of uncertainty and the likelihood of recovery for each assessment endpoint are described. Finally, conclusions are drawn for the entire ecosystem including all assessment endpoints.

9.1. SUMMARY OF RISKS TO THE ASSESSMENT ENDPOINTS

This section reviews the risks to the assessment endpoints by summarizing the lines of evidence and evaluating the likelihood of recovery.

9.1.1. Reproduction and Survival of Rainbow Trout

9.1.1.1. *Lines of Evidence*

The most critical factors limiting rainbow trout growth and survival include elevated water temperatures, irregular flows, and excessive sedimentation (Table 9-1). Lines of evidence supporting this conclusion include the frequency of exceeding standards for dissolved oxygen and temperature (Table 5-4), estimates of areas meeting habitat suitability indices (Figure 5-4), and field studies in the Middle Snake River and in other areas.

The frequency with which simulated dissolved oxygen falls below the standard for spawning for rainbow trout (Table 5-4) ranged from 0.19 to 0.58, i.e., from 19% to 58% of the time during the January 15 to July 15 spawning period. The frequency of exceeding the temperature standard for spawning was somewhat higher, ranging from 0.58 to 0.80, mainly because of a spawning season that extends into the summer months.

The risks to all life stages of rainbow trout from unacceptable habitat conditions were high throughout much of the Middle Snake River. These estimates of risk are based on habitat indices derived from water velocity, depth, and temperature and substrate type (Figure 5-4). Some areas with improved habitat are located just below the mouth of Rock Creek, in the reach between Kanaka Rapids and Crystal Springs, near Box Canyon, and below Bliss Dam.

Field work by Hill (1991c) demonstrated that spawning habitat for rainbow trout in some segments of the Middle Snake River has been adversely affected by sedimentation and high temperatures. Sedimentation clogs spawning gravels and reduces dissolved oxygen needed by developing embryos. A comparison to similar conditions in the Spokane River indicated that year-class strength in a native population of rainbow trout in the Spokane River was positively correlated with spring water flows.

Table 9-1. Factors limiting reproduction, growth, and survival of the rainbow trout population in the Middle Snake River

Factor	Stressor	LOE[a]	Risk	Uncertainty	Assumption	Recovery potential
Number of spawning fish	Loss of adult habitat (e.g., streamside vegetation, overhanging banks, and woody debris)	LIT, Field, BPJ	An increase in the population size is not possible with low or no reproduction	Low, field studies show low numbers of adult fish are present in the Middle Snake River	Lack of habitat is a main factor limiting the size of the adult population	Good, if habitat improvements can be improved, but low if a stable annual flow regime is not maintained
Spawning	Sedimentation, high water temperature, and land use on tributaries	HSI, WQS, LIT, Field, BPJ	Population of native fish cannot recover without successful reproduction	Moderate, historic spawning areas in the main channel have not been documented	Poor spawning success attributed to poor water quality conditions	Low, without improving water quality, reducing sedimentation, and controlling land/water use on tributaries
Rearing	Unstable stream flow during the spring and high water temperatures	HSI, WQS, LIT, Field, BPJ	Population cannot recover without successful recruitment	Rearing areas in the main stem of the Middle Snake River identified using habitat suitability indices	Rearing habitat important for maintaining and increasing adult populations	Low, the carrying capacity for native fish was likely permanently reduced by dam construction in the Middle Snake River
Overwintering	Loss of habitat (e.g., deep holes and large woody debris)	LIT, BPJ	Unknown	High, no information available on the amount of overwintering habitat in the main stem	Overwintering habitat can limit the size of the adult population	Unknown
Food supply	Sedimentation and increased water temperature	LIT, Field, BPJ	Poor growth, as invertebrate fauna do not support cold-water fish	Moderate, adequate analysis of sampling information has not been completed	Food supply can limit the size of a rainbow trout population	Low, without improving lotic conditions, lowering water temperature, and controlling sedimentation
Genetic diversity	Hybridization with stocked fish	LIT, BPJ	Poor survival with mixed genotype	Low, effects of hybridization on native fish are known	An adequate population of native fish remains	Good, provided that existing native fish are protected from stocked hatchery fish

[a]LOE - lines of evidence, HSI - habitat suitability indices, WQS - water quality standards, Field - field surveys in the Middle Snake River, LIT - literature, BPJ - best professional judgment.

Other factors that may affect native trout survival include competition from and hybridization with planted hatchery-reared rainbow trout.

9.1.1.2. *Likelihood of Recovery*

Without substantial changes in the management of flows in the Middle Snake River, the outlook for wild native rainbow trout appears bleak. Considering the habitat requirements discussed by Behnke (1992) and our estimation of the impairment of spawning, rearing, and adult habitat, the Middle Snake River cannot support a viable rainbow trout population. It is widely known that flow and water temperature have the most environmental influence on trout populations (Behnke, 1992), but management of the river to improve these conditions for these species has not occurred. Incremental improvement of rainbow trout habitat may be possible if potential spawning and rearing habitats in the river are identified, protected, and enhanced by water and land use management. The underlying question, however, is how many adult fish this system can support. In smaller rivers, native trout have returned and displaced warm-water species after their habitat was improved. Other studies (Partridge and Corsi, 1993; Lukas et al., 1995) found some suitable habitats in and near springs. However, these areas may not provide spawning and rearing habitat at levels that would sustain a population of native rainbow trout in the Middle Snake River.

9.1.2. Reproduction and Survival of the Mountain Whitefish

9.1.2.1. *Lines of Evidence*

The most important factors limiting the recovery of the mountain whitefish population in the Middle Snake River appear to be high water temperatures, loss of lotic habitat, fluctuating water flows, and sedimentation (Table 9-2). The evidence for these effects is demonstrated by evaluating favorable spawning conditions (<7°C during a 75-day spawning-incubation window) at two locations in the river from 1970 to 1994 (see p. 8-6, Effects on Spawning Activities), comparison with habitat suitability indices (Figure 5-5), and review of the literature.

The evaluation of water temperatures during the spawning season at two locations, above and below the Thousand Springs Complex, showed that favorable spawning conditions did not occur from 1970 to 1994, whereas the frequency with which simulated temperature exceeded the standard (no greater than 9°C) for spawning for mountain whitefish (Table 5-4) ranged from 0.05 to 0.22 (5% to 22% of the time) during the spawning season. The implication here is that water quality standards may not be adequate to protect the optimum spawning requirements for this fish.

Table 9-2. Factors limiting reproduction, growth, and survival of the mountain whitefish population in the Middle Snake River

Factor	Stressor	LOE[a]	Risk	Uncertainty	Assumption	Recovery Potential
Number of spawning fish	Loss of adult habitat, restrictions on movement to feeding and overwinter areas	LIT, BPJ	An increase in the population size is not possible with low or no reproduction	Low, very few adult fish are present in the Middle Snake River or nearby tributaries	Available fish surveys are a true indication of the scarcity of fish in this area	Low, without improving lotic conditions and lowering water temperature
Spawning	Water temperature too high for successful egg development, loss of lotic habitat	HSI, LIT, WQS, BPJ	Population cannot recover without successful reproduction	Low, water temperature requirements for spawning are known	Low spawning success attributed to poor water quality conditions	Low, the present annual water temperature regime does not support successful spawning
Fry	High water temperature, fluctuating water levels, loss of lotic habitat, and predation	HSI, LIT, WQS, BPJ	Population cannot recover without successful recruitment	Low, temperature requirements are known, but rearing habitat not identified	Fluctuating water levels push fry into deeper water where increased predation occurs	Low, the present annual water temperature regime does not support successful fry development
Food supply	Sedimentation and increased water temperature	LIT, Field, BPJ	Invertebrate fauna not adequate to support an increased mountain whitefish population	Moderate, adequate analysis of sampling information has not been completed	Food supply is limiting	Low, without improving lotic conditions and lowering water temperature
Movement	Dams prohibit seasonal migratory movement by adults	LIT, Field, BPJ	Adults unable to reach upstream tributaries or refugia used for spawning or rearing	Moderate, limited information on migratory population in the Middle Snake River	Adult fish reported at Lower Salmon Falls Dam in the 1950s represented a migratory population	Low, without providing adequate fish passage facilities at the dams

[a]LOE - lines of evidence, HSI - habitat suitability indices, WQS - water quality standards, Field - field surveys in the Middle Snake River, LIT - literature, BPJ - best professional judgment.

The evaluation of the frequency with which simulated dissolved oxygen exceeded water quality standards (>90% saturation) during the spawning season for this fish ranged only from 0.01 to 0.11 (Table 5-4). Low dissolved oxygen is not an issue because these fish spawn during the winter when saturation levels for dissolved oxygen are high.

The risk estimates for mountain whitefish based on habitat suitability factors (water velocity, depth, and temperature and substrate type) show high impairment for all life stages (Figure 5-5). The least impairment occurred for the adult stage in the reach from Thousand Springs to Lower Salmon Falls. This may be due to variation in the sources of water to the river, where intrusions of cool, clear water alleviate the adverse effect of low flow and increased sedimentation.

Other factors such as competition with other salmonid species for any available habitat left in the Middle Snake River do not appear to limit mountain whitefish populations. Across their range in North America, mountain whitefish coexist and may compete with several other fish species.

9.1.2.2. ***Likelihood of Recovery***

Similar to the native rainbow trout, the recovery potential for mountain whitefish in the Middle Snake River is low. Mountain whitefish in this area face serious habitat deficiencies, including poor spawning and rearing conditions, reduction and alteration of food sources, and loss of free movement to feeding and overwintering areas. The loss of lotic habitat, fluctuating water levels, poor water quality/quantity, and restriction of fish movement are associated with dam construction and operation. Some of these impacts may be lessened by changing the way the dams are operated, but there appears to have been a net loss of habitats for all life stages of the mountain whitefish that cannot be mitigated.

Some spawning and rearing habitat in adjacent tributaries has also been altered or destroyed by land and water use operations and activities. These tributaries provide valuable refuge for mountain whitefish and need to be protected to maintain existing populations and for any possible recovery.

Other factors, in combination with previously discussed impacts, may also contribute to the low numbers of mountain whitefish in the study reach. For example, this reach is below the elevation range (1,370 to 2,225 m) where this fish is usually found in other parts of its range in the western United States.

Table 9-3. Factors limiting reproduction, growth, and survival of the white sturgeon population in the Middle Snake River

Factor	Stressor	LOE[a]	Risk	Uncertainty	Assumptions	Recovery potential
Number of spawning fish	Overharvesting	Field, LIT, BPJ	Low numbers of adult fish available to spawn; size of spawning populations will decline	Low, but information on the size of the population is limited	Available population censuses are accurate	Fair, if spawning conditions and recruitment can be improved
Spawning activity	Low flows and high water temperature	HSI, WQS, Field, LIT, BPJ	Population cannot be maintained or increased without successful reproduction	Low, water conditions needed for spawning are known	Low spawning success attributed only to poor water conditions	Good, if spring flows are adequate to trigger spawning in Apr-May
Larval development	High water temperature	HSI, WQS, Field, LIT, BPJ	Window for larval development too small, recruitment to population declines	Low, water temperatures needed for incubation are known	A 6-week incubation window is needed for adequate recruitment	Good, if water temperatures can be held below 18°C during a 6-week incubation window
Predation on eggs and larvae	Loss of dynamic lotic habitat	LIT, BPJ	Eggs will not be widely dispersed in low flows and will be susceptible to predation, disease, and fungus	Moderate, because of limited information on actual loss due to predation, disease, and fungus	Loss of dynamic lotic habitat provides favorable conditions for egg predators	Good, if > 15,000 cfs flows can be maintained during and 6 weeks following spawning
Rearing habitat	Dams caused a 37% reduction of free-flowing habitat and increased sedimentation	HSI, WQS, Field, LIT, BPJ	Rearing habitat will continue to decline, resulting in a further reduction in the carrying capacity. Rearing habitat not large enough to handle planted fish	Moderate, the amount of rearing habitat has not been documented in the Middle Snake River. The largest population of white sturgeon in this area resides in a reservoir	Rearing habitat is limited	Low, the carrying capacity of the Middle Snake River for white sturgeon was permanently reduced with dam construction, but the remaining habitat can be protected by proper management of flows and land uses

Table 9-3. Factors limiting reproduction, growth, and survival of the white sturgeon population in the Middle Snake River (continued)

Factor	Stressor	LOE[a]	Risk	Uncertainty	Assumptions	Recovery potential
Mortality on: Young-of-the-year	Predation and contaminant exposure	LIT, BPJ	Low year-class survival; recruitment will not maintain population	High, very little information available on the causes of mortality	Available population censuses are accurate	Unknown, there is incomplete information on factors causing mortality
Juveniles	Passage through dams	LIT, BPJ	Passage through or over dams causes unacceptable mortalities	High, actual mortality rates from passage of hatchery fish are unknown	Wild fish also move downstream through and over dams	Low, without installation of proven fish bypass facilities at each dam
Adults	Illegal harvest	LIT, BPJ	Loss of potential spawners in the recreational fishery	No information on the illegal take of sturgeon	Some fishermen need or take sturgeon for food	Fair, illegal take can be minimized with a public awareness program
Food supply	Reduced food supply, loss of anadromous salmonids	Field, LIT, BPJ	Food chain leading to white sturgeon is less complex and more susceptible	White sturgeon, an opportunistic feeder, can develop an alternative prey base	Main-stem dams in and above Hells Canyon will not be removed	Low, carrying capacity was reduced by the loss of salmonids and lampreys from the food chain
Toxics	Chemical contamination of food chain (none observed to date)	BPJ	Low fecundity and reduced survival	High, information on the total load of contaminants entering the food chain leading to white sturgeon is limited	Long-lived fish such as white sturgeon will bioaccumulate contaminants	Low, the Middle Snake River is a sink for chemical contaminants used in the watershed
Genetic diversity	Isolated small populations	LIT, BPJ	Reduced overall health of populations	The amount of time needed to see effects is unknown and maybe in the hundreds of years	Isolation causes a reduction in genetic diversity	High, brood stock could be introduced from the lower Snake River

[a]LOE - lines of evidence, HSI - habitat suitability indices, WQS - water quality standards, Field - field surveys in Snake River, LIT - literature, BPJ =best professional judgment.

9.1.3. Reproduction and Survival of the White Sturgeon

9.1.3.1. *Lines of Evidence*

Factors limiting the growth and survival of the white sturgeon population in the Middle Snake River are characterized in Table 9-3. The main factors affecting white sturgeon appear to be unfavorable spawning and incubation conditions due to high water temperatures and/or low flows, the loss of spawning and rearing habitat, and low recruitment owing to a low number of spawning fish in some reaches. It is also possible there is increased egg-larval stage mortality due to predation and disease. The evidence linking these factors to white sturgeon growth and survival are the habitat suitability indices (Figure 5-6), frequency of exceeding temperature standards (Table 5-4), and field and laboratory studies reported in the literature.

Risk estimates based on habitat suitability indices showed the lowest survival for life stages from spawning through larval development in the reach above Lower Salmon Falls Dam (Figure 5-6). In comparison, juvenile and adult habitat has the lowest impairment from the Box Canyon reach downstream to King Hill. An analysis of habitat suitability indices was not completed for the reach from King Hill to C.J. Strike, where the main white sturgeon population resides.

The frequency that simulated values of water temperature exceeded Idaho's water quality standards (for cold-water biota, Table 5-4) ranged from 0.01 to 0.20 (i.e., up to 20% of the time during the year). The river areas having greater exceedances were from Milner Dam to Kanaka Rapids and below Bliss Dam (Table 5-4). It is important to note that the water quality standard used in this analysis, a maximum daily average of no greater than 19°C, falls in the range (18°C-20°C) where substantial mortality occurs to the embryonic stages of the white sturgeon (Wang et al., 1985, 1987). Therefore, the risk of adverse effects to the developing stages of white sturgeon is likely to occur much more than 20% of the time. Similar to whitefish, the water quality standard applied in the Middle Snake River does not appear to be adequate to protect the spawning requirements of the white sturgeon.

Other factors listed in Table 9-3, with more information, may prove to be more important than presently considered. For example, Chandler and Lepla (1997) believed that spawning activity immediately below C.J. Strike Dam was potentially adversely affected by daily block loading at the dam. Block loading at C. J. Strike Dam occurs when the dam is operated to follow peak power loads. This generally results in a sudden increase followed by a sudden decrease in water flows in the morning and evening each day. The change in water flow associated with block loading effectively reduces spawning habitat for the white sturgeon. It is not known whether block loading at Bliss Dam has any adverse effects on white sturgeon spawning habitat in the King Hill reach.

It is unknown how many of the approximately 5,200 white sturgeon planted in the Middle Snake River between C.J. Strike Dam and Shoshone Falls will survive and reproduce. The overall success of these fish plants has not been determined. Observations by Lepla and Chandler (1995a,b) showed that hatchery fish stocked below Lower Salmon Falls Dam were entrained through Bliss Dam, but with unknown turbine mortality. It also appears that hatchery fish may not be adjusting to the conditions in the Middle Snake River, as noted by their significantly lower condition factors when compared with similarly sized wild fish in the Bliss reach (Lepla and Chandler, 1995a).

Factors other than those discussed above may also be affecting the growth and survival of the white sturgeon population in the Middle Snake River. These factors include mortality of juvenile sturgeon passing over or through the turbines at dams, disease, and exposure to riverborne contaminants.

9.1.3.2. *Likelihood of Recovery*

If the white sturgeon populations in the Middle Snake River are to be restored to levels capable of sustaining some level of recreational harvest, habitat and flow restoration will be required. White sturgeon evolved to depend on unrestricted spring freshets on an annual basis as spawning cues. Recovery would require difficult choices regarding restoration of flows for long-term spawning and rearing success. Flow management is possible, as the water temperatures and flows needed for successful spawning are generally known. Maintaining these conditions will require further information on the survival of year classes reared under different or poorer conditions. For example, some spawning activity was observed in 1993 in the Bliss reach at flows less than 283 m^3/s (10,000 cfs), but it was followed by an incubation window with water temperatures unfavorable for larvae survival. Further, potential and existing rearing areas for white sturgeon need to be identified in each reach. If recovery is to be successful, the rearing areas need to be enhanced by restoration of spring freshets. These areas also need to be protected by land and water use planning on a long-term basis.

9.1.4. Reproduction, Survival, and Diversity of Macroinvertebrates

9.1.4.1. *Lines of Evidence*

The lotic invertebrate fauna of the Middle Snake River have been affected by dam development for hydroelectric power and irrigation, flow reduction, water quality deterioration, and the presence of the New Zealand mudsnail (*Potamopyrgus antipodarum*), an exotic species.

In general, the endemic mollusc populations in the Middle Snake River appear to have declined from historical abundances and have become more localized in distribution. It is clear that no single river habitat favors the existence of all the species. For example, the Snake River

Physa requires deep, swiftly flowing water whereas the Utah valvata inhabits pools and macrophyte beds with sediments of sand and silt (Figure 9-1). Declines in mollusc populations are likely due to loss of much of the original habitat heterogeneity in the Middle Snake River following multiple impoundments of the Snake River. Historically, this heterogeneity facilitated the co-occurrence of molluscs with very disparate habitat requirements. Once-common river habitat features in the Middle Snake River, such as large rapids and cascades, are now rare habitat "islands" often separated by nearly lentic environments.

The probability of exceeding limits of temperature (<22°C, or maximum daily average of 19°C) for cold-water biota (Table 5-4) is an indication of the areas in the Snake River that are not likely to support the growth and survival of native cold-water macroinvertebrates. The current state of the macroinvertebrate community itself is evidence that the temperature needs of native macroinvertebrate communities are not being met.

9.1.4.2. *Likelihood of Recovery*

Improvement of the macroinvertebrate community (e.g., greater taxa richness, the presence of pollution-intolerant taxa, and sustainable populations of endemic molluscs) is unlikely to occur without substantial improvement of in-stream habitat conditions. Such improvements include reduced water temperature, reduced sediment loads, and increased water velocities (Tables 9-4 and 9-5).

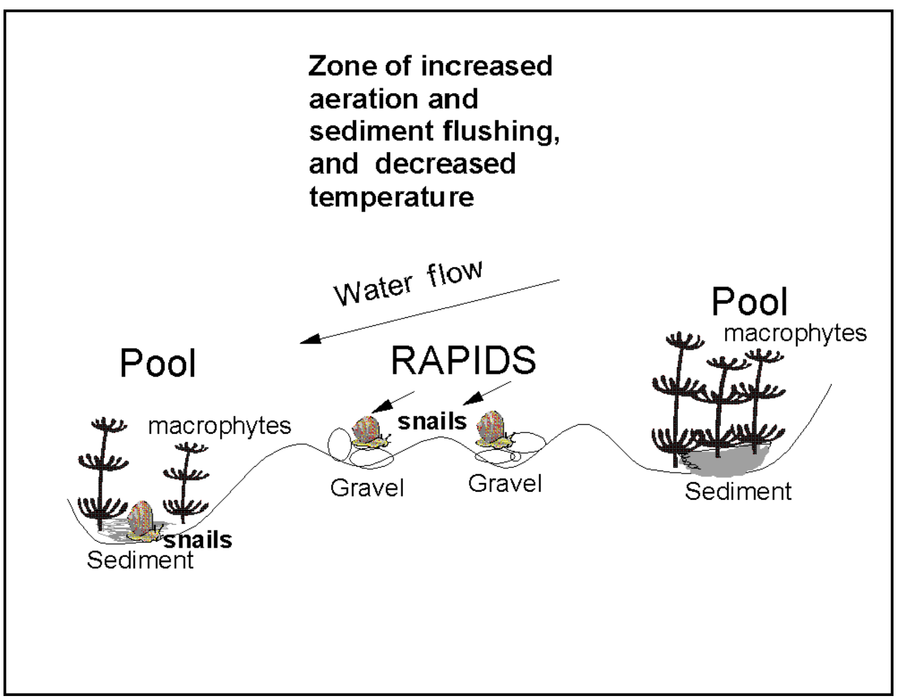

Figure 9-1. Factors controlling molluscs survival in the Middle Snake River.

Table 9-4. Factors limiting the reproduction, growth, and survival of macroinvertebrates in the Middle Snake River

Life stage	Stressor	LOE[a]	Risk	Uncertainty	Assumptions	Recovery potential
Eggs	Temperature	WQS, LIT, BPJ	Thermal stress if maximum tolerances exceeded	No single river habitat favors the existence of all species Observations of sedimentation are limited to one very high-flow year and a succession of low-flow years	Nearly all of the adverse effects are caused by deteriorated water quality (increased temperature and sedimentation), reduced flows, and the loss of habitat	Unlikely without substantial improvements in water quality and stream flows
	Sedimentation	Field, LIT, BPJ	Burial and scouring			
Larvae/ Nymphs	Temperature	WQS, LIT, BPJ	Thermal stress if maximum tolerances exceeded			
	Sedimentation	Field, LIT, BPJ	Loss of large, stable substrate; burial of food resources			
	Turbidity	LIT, BPJ	Loss of diatoms (food) and reduced visibility for predators or other visual feeders			
	Water velocity (flow)	Field, LIT, BPJ	Loss of lotic habitats; increased sedimentation and water temperatures			
	Riparian vegetation	LIT, BPJ	Loss of organic material (food)			
	Exotic species	Field, LIT, BPJ	Potential for competition should native species recover			

[a]LOE-lines of evidence, HSI-habitat suitability indices, WQS-water quality standards, Field-field surveys Snake, LIT-literature, BPJ-best professional judgment.

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Table 9-5. Factors limiting the reproduction, growth, and survival of aquatic molluscs in the Middle Snake River

Life stage	Stressor	LOE[a]	Risk	Uncertainty	Assumptions	Recovery potential
Eggs	Temperature	WQS, LIT, BPJ	Thermal stress if maximum tolerances exceeded	No single river habitat favors the existence of all species	Nearly all of the adverse effects are caused by deteriorated water quality (increased temperature and sedimentation), reduced flows, and the loss of habitat	Unlikely without substantial improvements in water quality and stream flows
	Sedimentation	LIT, BPJ	Burial and scouring			
Juveniles and adults	Temperature	WQS, LIT, BPJ	Thermal stress if maximum tolerances exceeded	Observations of sedimentation are limited to one very high-flow year and a succession of low-flow years		
	Sedimentation	LIT, BPJ	Loss of large, stable substrate; burial of food resources			
	Water velocity (flow)	LIT, BPJ	Loss of lotic habitats; increased sedimentation and water temperature			
	Exotic species	LIT, BPJ	Potential for competition should native species recover			
	Isolated populations	BPJ	Might result in a loss of genetic diversity. Many populations currently limited to relatively small areas of refuge habitats. Loss of refugia could lead to local or complete extinction.			

[a]LOE-lines of evidence, HSI-habitat suitability indices, WQS-water quality standards, Field-field surveys in Snake, LIT-literature, BPJ-best professional judgment.

It appears that several tributaries to the Middle Snake, as well as some downstream locations, contain species of aquatic insects not currently found in the Middle Snake River. Should in-stream conditions improve within the Middle Snake River, these outside populations would likely serve as a source of colonists for the Middle Snake River. Thus, given improvement in the water quality and benthic habitat of the Middle Snake, the potential for recovery of aquatic insects is high.

The recovery of mollusc populations also is dependent upon improved habitat conditions in the Middle Snake River. However, the recovery potential for these organisms is not as great as that for aquatic insects because:

- dispersal, particularly in an upstream direction, is limited;
- source populations for molluscs are not as large or widespread as those for aquatic insects; and
- genetic diversity within the source populations may have been reduced.

9.1.5. Growth and Diversity of Phytoplankton, Macrophytes, and Epiphytes

9.1.5.1. *Lines of Evidence*

9.1.5.1.1. *Phytoplankton.* Plankton chlorophyll *a* values would classify the Middle Snake River as eutrophic when rooted plants are low and meso-eutrophic when rooted plants are dense. Sediment nutrients, dissolved nutrient levels, and aquatic plant biomass all indicate that the river is more eutrophic than indicated by the phytoplankton chlorophyll *a* levels.

9.1.5.1.2. *Macrophytes and epiphytes.* The risk that total macrophyte and epiphyte biomass will exceed 200 g/m^2, established as the standard for this study, is high at the Crystal Springs to Boulder Rapids reach on the basis of the simulation. A review of the literature supports this risk estimate. Densities measured in the Crystal Springs, Box Canyon, and Thousand Springs reaches all compared with, or exceeded, levels measured in other systems classified as eutrophic (Nichols and Shaw, 1986; Kennedy and Gunkel, 1987; Anderson and Kalff, 1988; van Wijk, 1988; Falter et al., 1991; Urbanc-Bercic and Glejec, 1993). The evidence that nutrients are high is also substantiated by simulations that show that the likelihood of exceeding a phosphorus limit of 0.075 mg/L is extremely high between Rock Creek and Crystal Springs because of the total phosphorus load from the City of Twin Falls STP, fish hatcheries, and irrigation returns (Figure 5-1). Again, as with other water quality parameters, the nutrient concentrations decrease below Bliss Dam because of spring inflows (Figure 5-2).

These exceptional growths were possible because of sedimentation and shallowing of the water depth along with concomitant nutrient enrichment of sediments and the water column. Once that physical/chemical set of precursor conditions was in place, a low-flow year such as

1992 resulted in very high levels of aquatic plant development, in particular heavy growths of epiphytes. Slightly higher summer flows in 1993 and 1994 altered macrophyte development, especially by reducing early-season epiphyte development.

As in most water bodies, the dominant controlling factors of the attached plant community in the Middle Snake River are water depth and clarity (both controlling available light), substrate composition, substrate nutrient content, and water velocity. Ideal combinations of these dominant controlling factors are shallow, clear, slow waters over fine, nutrient-rich sediments (Table 9-6). Given ideal conditions of these factors, nutrient concentrations of overlying water may further enhance plant communities, especially nonrooted forms.

The majority of plant biomass in the Middle Snake River is found in water less than 2 m in depth. Except for the higher velocity rapids and narrow intra-plant-bed open channels, water velocities in the Middle Snake River are generally less than 1 m/sec, within the optimal velocity range for submerged aquatic macrophytes (Haslam, 1978; Biggs, 1996). Alcove springs (springs discharging from the lower canyon walls along the Snake River banks) are common along the Middle Snake River below Twin Falls. These springs discharge about 170 m^3/s (6,000 cfs) year-round to the river. Exceptionally clear waters (Secchi disk transparency $\geq$ 10 m) and cool water of the springs influence the river channel for a distance downstream of the springs. This combination of a constant water environment and the increased transparency of the water downstream of the springs/river confluences provides optimal conditions for attached plant growth.

The process of macrophyte development begins in sediments that are predominantly fine sands and silts. These sediments cause shallowing of extensive reaches until water depths approach approximately 2 m. At that point, sufficient light reaches the bottom to support macrophyte growth (Figure 9-2). If the sediment organic load is too high, macrophyte growth will be inhibited. As the density of the plant bed increases, more sediments are trapped along with nutrients.

Epiphyte biomass is later incorporated into sediments, supplementing nutrient levels at a time when rooted plant uptake has started to deplete sediment nutrients from the root zone. Epiphytic algae cannot reach high biomass levels until three required conditions are met: (1)high enough temperatures (Millner et al., 1982), (2) sufficient nutrients from the water column, and (3) adequate physical support provided by rooted plants high enough in the water column to obtain their required high light nearer the water surface. By midsummer, the combined community of epiphytes and nonrooted macrophytes apparently outcompetes rooted forms.

Table 9-6. Factors limiting reproduction, growth, and survival of aquatic plant communities in the Middle Snake River

Life stage	Stressor or driving force	Season	LOE[a]	Risk	Uncertainty	Assumptions	Recovery potential
Rooted macrophytes	Deep fine rich sediment		Field	Eutrophication enhanced growth, highest densities were below agricultural discharges, no other sources in these areas	Distribution of stressors Observations of sedimentation are limited to one very high flow year and a succession of low-flow years	A level of plant biomass is an integral component of this aquatic ecosystem.	Spring communities are more diverse and balanced
	Organic matter in sediment		Field	Eutrophication enhanced growth			Sediment removal
	Nutrients in sediment		Field	Eutrophication enhanced growth			
	Water velocity slowed		Field	Eutrophication enhanced growth			
	Low flow		Field	Eutrophication enhanced growth			
	Cool temperature	Spring intermittent	WQS	Epiphyte growth 1993-94			
	Light suppression		LIT, BPJ				
	Total phosphorus		Field, LIT				
Nonrooted macrophytes and epiphytes	High temperature physical support (macrophytes) and dissolved nutrients	Late summer and fall	Field	Eutrophication; suppression of macrophytes	Distribution of stressors	A level of plant biomass is an integral component of this aquatic ecosystem.	Sediment removal
	Phosphorus > 1,100 mg/g		Field	Decline in biomass			Water column nutrient reduction
	Dissolved oxygen < 6 mg/L, pH > 9, and high TOC		Field	No documented impact on ecosystem			
Phytoplankton	Nutrients		Field	High productivity		A level of phytoplankton chlorophyll a is in balance with the other biological components of this ecosystem.	Water column nutrient reduction

[a]LOE-lines of evidence, HSI-habitat suitability indices, WQS-water quality standards, Field-field surveys in the Middle Snake River, LIT-literature, BPJ-best professional judgment.

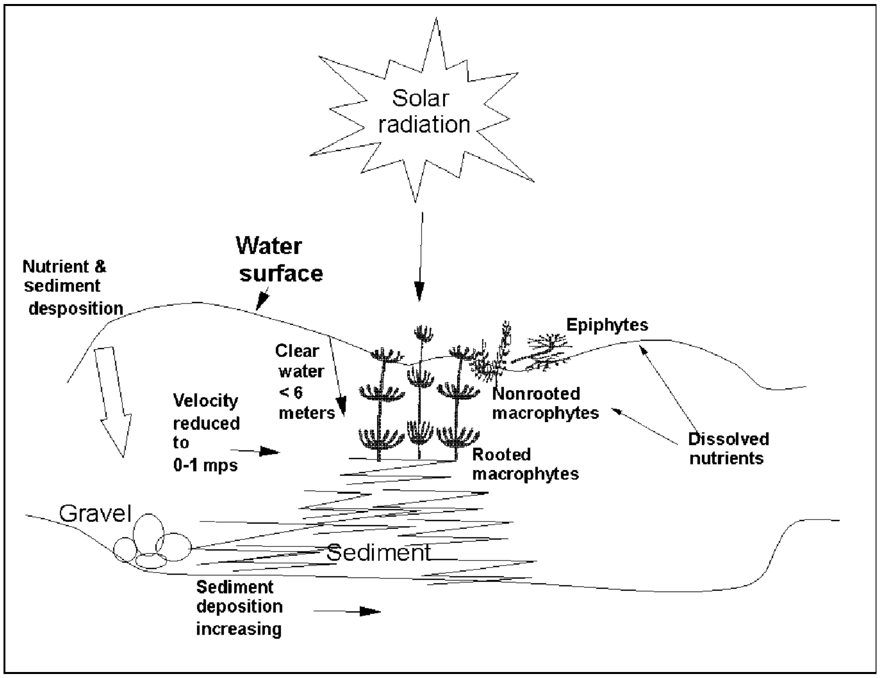

Figure 9-2. Factors controlling aquatic plants in the Middle Snake River.

By including *Ceratophyllum demersum* with the epiphyte component, a strong relationship was identified between the epiphytic portion of total aquatic macrophyte biomass and water velocity. This grouping was based on the fact that *Ceratophyllum demersum* does not have a well-developed root structure and obtains its nutrients in a similar fashion to filamentous epiphytic algae, i.e., from the water column (Kennedy and Gunkel, 1987).

9.1.5.2. *Likelihood of Recovery*

Growth of macrophytes as simulated by the model (Figure 5-3) is predicted to exceed "nuisance" levels, characterized by a maximum biomass greater than 200 mg/m^2 (ash-free dry matter). Such growth is highly probable, especially in low-flow years, as long as current management practices continue.

The likelihood of recovery is dependent upon the availability of clear, cold water, with high flows capable of scouring out long-deposited sediments. Spring water is one source of clear, cold water. The composition of aquatic plant communities in the alcove springs along the Middle Snake River is more diverse and balanced than in the communities in the main channel of the Middle Snake River. This is an indication of the type of plant community that the river would support if the excess levels of nutrients and sediments were diminished and flow patterns followed the natural hydrograph.

As to the eventual outcome of continued enrichment, Moss (1976) concurred with the general observations of sewage and fish pond managers that continued nutrient enrichment to hypereutrophic ranges of limiting nutrients will push the community through rooted macrophyte dominance and epiphyte dominance, attaining phytoplankton dominance with eventual complete suppression of the macrophyte community.

9.2. SOURCES OF UNCERTAINTY

One of the goals of ecological risk assessment is to describe all sources of uncertainty. Sources of uncertainty in this assessment stem from the simulation model and qualitative analysis of effects due primarily to the lack of knowledge about the specific life histories of species in the Middle Snake River. For the qualitative analysis, there is limited information on the native macroinvertebrate species. Without site-specific data, the analysis is based on similar species, genera, or even members of the mollusc family. This information is evaluated by scientists with expertise in the field of macroinvertebrate ecology. Their judgment is biased according to their experience and education. In addition, their analyses often include a review of field studies where measurement error can bias the results.

A mathematical model was used to simulate ecosystem dynamics in the Middle Snake River from Rkm 1,028 to Rkm 877.6 (RM 640 to 545.5) for the period from January 1, 1990, to December 31, 1994. As part of the uncertainty analysis, surface water quality results were compared with water quality collected at various locations in the river by the University of Idaho's Agricultural Research Station, Idaho State University, and Clear Springs Foods, Inc. The macrophyte biomass estimates generated by the model were compared to observations made by the University of Idaho (Falter and Carlso, 1994; Falter et al., 1995; Falter and Burris, 1996).

The sources of uncertainty in the simulation model are presented in the following categories:

- ecosystem driving forces and stresses,
- uncertainty in sources of mass and energy,
- model error,
- parameter estimation error,
- measurement error,
- quantitative measures of exposure and effect, and
- lack of knowledge.

A detailed discussion of the uncertainties in the simulation is provided in Appendix D of this report. The following is a brief summary of that discussion.

9.2.1. Variability in Driving Forces and Stresses

Principal environmental factors for the study reach are hydrology and meteorology. The conceptual model accounts for variability and uncertainty of these environmental factors by assuming that the 67-year data record from 1928 to 1994 for river flow and air temperature for Twin Falls and Glenns Ferry is a representative sample for hydrology and meteorology.

The models for variability in both hydrology and meteorology are based on the actual 67-year record in the case of the meteorology and the adjusted 67-year record for the hydrology. Although using the actual record captures some of the variability in a simple and straightforward manner, it represents only a sample of the actual population. As such, it may not include events that have a low probability of occurrence. Such events would likely be associated with long-term changes in regional and global weather patterns.

Because of changes in the hydrologic regime, the existing record of actual flows cannot be used directly to characterize risk associated with present management of the system hydrology. However, the Idaho Department of Water Resources (IDWR) has developed a model that estimates the historical monthly average flow and reach gain in the Snake River given present-day operating rules for the system. There is uncertainty in the way in which the monthly flows estimated by the IDWR hydrologic model were disaggregated into daily values. The results of applying the model to the period 1928-1994, reported as IDWR Study 150 (Robert Suter, IDWR, personal communication), are used to characterize hydrologic variability for the risk analysis in the study reach.

9.2.2. Sources of Mass and Energy

Correlation of modeled results with observed values for water quality was generally high with a few exceptions. The correlation between simulated and observed values of total ammonia nitrogen and total phosphorus was generally low. The total ammonia nitrogen results showed significant positive bias (the model predicted higher values, on the average, than were observed) downstream from the City of Twin Falls Sewage Treatment Plant (STP) and Auger Falls and in the area of dense macrophyte beds. The bias in the total ammonia nitrogen at the station downstream from Auger Falls and from the City of Twin Falls STP could be due to error in the loading from the City of Twin Falls STP. It could also be due to incorrectly characterizing the sources and sinks for total ammonia in the Auger Falls segment of the study reach.

With the limited data available, it was difficult to account for or separate uncertainty in the sources of mass and energy. The longest record available for developing models for the sources of mass and energy was 5 years (1990 to 1994). In addition, sampling periods for these sources were generally biweekly or greater. In the case of the springs, which play an extremely important role in the study reach, data gaps were much larger. The relatively low variability of both the quantity and quality of the spring flows mitigates the impact of these data gaps to a degree. For sources with important high-frequency components such as irrigation return flows, the data gaps are likely to be significant. In general, the limited data imply a high degree of uncertainty for both high-frequency and low-frequency components of the sources of mass and energy.

9.2.3. Model Error

Errors in the simulation model for ecological risk analysis contribute to uncertainty in the estimate of ecological risk in a number of ways. Errors in structure are generally the major sources of uncertainty in ecological models. Structural errors result from omitting important variables or flow paths between variables.

Spatial and temporal aggregation of variables in the simulation model also contribute to model uncertainty. The equations of mass and energy balance for this model assume the variables describing the water body vary only vertically or longitudinally, depending on the nature of the water body. In addition, the simulated variables represent volume-averages for river segments 0.5 to 5 km (0.3 to 3.3 miles) in length and reservoir segments 1.5 meters (5 ft) thick. Variables are also averaged over a 3-hour period.

The model structure may not include all the processes required to accurately simulate primary productivity in the study reach. For instance, correlation between simulated and observed concentrations of dissolved oxygen is generally good, with the exception of the first part of 1992, when supersaturated levels of dissolved oxygen were observed (Minshall et al., 1993) but were not simulated by the model.

The conceptual model for the macrophytes includes the assumption that nutrient flow from the sediments to the roots of macrophytes is unlimited by plant uptake. This assumption plays a significant role in the development of management strategies for reducing macrophytes in the study reach. Given an essentially infinite supply of nutrients, reduction in the discharge of dissolved nutrients is unlikely to result in a decrease in macrophyte biomass. Although results of macrophyte studies from 1990 to 1994 in the study reach (Falter and Carlson, 1994; Falter et al., 1995; Falter and Burris, 1996) suggest that the assumption may not be met in the densest plant beds in late summer (i.e., some phosphorus depletion from sediments was observed), it is a reasonable first approximation for extended periods of low flow when sediment deposition rates are high. However, under conditions of high flow, when sediments are being removed by scouring, it is likely to result in unreasonably high rates of nutrient flow to rooted macrophytes.

Structural errors in the model also may be an important source of bias and poor correlation between simulated and observed total ammonia nitrogen in the vicinity of the macrophyte beds at Crystal Springs and Boulder Rapids. This segment of the study reach is one of high biological productivity. Uptake and release of ammonia by macrophytes is an important part of the nutrient dynamics in the model. Additional field and/or laboratory studies would be required to adequately test the hypotheses that control ammonia uptake and release in the model.

The model predictions for macrophytes and epiphyte growth in the Crystal Spring reach and Box Canyon reach were generally correct. There were some exceptions. In 1992, the predicted rooted and nonrooted macrophyte densities in Crystal Springs reach were about 3 to 4 times greater than observed values. Also, the observed epiphyte demise preceded the predicted timing. A low-flow year such as 1992 resulted in very high levels of aquatic plant development, in particular, heavy growths of epiphytes. It is likely that the warm, low flows of 1992 favored early epiphyte development which suppressed rooted and nonrooted macrophytes by blanketing, light suppression, and nutrient competition through the first half of the summer. Slightly higher summer flows of 1993 and 1994 altered macrophyte development, especially by reducing early-season epiphyte development. In 1993, predicted rooted macrophytes closely modeled observed, both in magnitude and timing. Predicted nonrooted macrophytes were slightly lower and about a month later than observed. These biological interactions were not incorporated into the model.

The model was also quite sensitive to the magnitude of the habitat factors. The habitat factors, in principle, should be derived from knowledge of sedimentation processes. However, in this case, lack of knowledge of these processes made it necessary to use the limited data that were available (Hill, 1992) regarding macrophyte habitat. Although these data provided information about conditions existing during 1990 for certain segments of the study reach, they were not sufficient for predicting habitat conditions under various river flow regimes. Recent data (McLaren, 1998) show that habitat factors for macrophytes change dramatically as a result of scouring of sediment deposits and associated macrophytes during periods of high flows in the Snake River. On the basis of his study of sediment transport during July and October 1997, McLaren (1998) concluded habitat factors change when the flow in the study reach exceeds approximately 283 m^3/s (10,000 cfs). Such conditions have occurred during the past, but are not reproduced in the cumulative distribution function generated by the model. Incorporating a predictive model for macrophyte habitat would require more study of the sedimentation processes in the study reach.

The simulation results for macrophytes were also sensitive to the parameters used to characterize the physical stress placed on plants by high water velocities. This component of the model had much greater impact on macrophyte density than did changes in the concentration of total phosphorus in the water column. The initial estimates of the parameters used to characterize physical stress from high water velocity were estimated from the work of Chambers et al. (1991a,b). The magnitude of parameters relating macrophyte growth to water velocity was modified to account for the results obtained by McLaren (1998). However, this change was not able to account for the dramatic changes in macrophyte growth observed in 1997, which were likely due to the fact that these parameter changes affected only the plant physiology and were not related to the physical processes associated with sediment transport.

As is the case for most ecological models, the structure for the ecological model used in this analysis of risk is a hypothesis derived from previous ecological model construction and field studies in rivers, lakes, and reservoirs, including field studies done in the study reach. There are, at this point, no widely accepted protocols for testing hypotheses regarding state-space models of the type used in this analysis. Oreskes et al. (1994) suggest a qualitative comparison of model simulations with observed values of the variables as a way of establishing the credibility of earth science models.

9.2.4. Parameter Estimation

Formal solutions of the parameter estimation problem are difficult to obtain. The difficulty arises from the nonlinear nature of the mass and energy balance equations, limited data, and the large number of parameters. Because of this, parameter estimation was done by trial and error for selected parameters only. The trial-and-error process included initial selection from the literature, followed by adjusting parameters until simulated values agreed reasonably well with observations.

Those parameters included in the trial-and-error process were ones that determine the sedimentation rates of phosphorus: loss, uptake, and release of ammonia nitrogen; growth and death rates of macrophytes and epiphytes; and habitat factors for macrophytes. Model response was particularly sensitive to the parameters for growth and mortality rates of macrophytes, epiphytes, and habitat factors for macrophytes. Of particular importance for parameters characterizing growth rates and mortality rates of macrophytes and epiphytes was the role of the sediments and sedimentation processes in the study reach.

9.2.5. Measurements

Field measurements of variables are needed for testing model hypotheses and ultimately for evaluating the reliability of the simulation methods. However, there are several sources of uncertainty in field measurements. There is generally some error associated with the instrument or laboratory technique and with the manner in which samples are taken and handled. In general, the largest variability is usually due to spatial or temporal variability of the variable being measured. This is particularly true for the biological variables. For example, Falter et al. (1995) and Falter and Burris (1996) observed high spatial variability of macrophyte density at Boulder Rapids with a high number of replicates. Dissolved oxygen is a variable that can also have high temporal variability during periods of high primary productivity. There will be uncertainty in characterizing the average state of dissolved oxygen in the system when only one grab sample is taken every 2 weeks.

There is also uncertainty in comparing field observations with simulated results because the field observations are point measurements whereas the simulated values are space- and time-averaged. The magnitude of the uncertainty depends, of course, on each variable's spatial and temporal characteristics.

9.2.6. Quantitative Measures of Effect

Uncertainty in the quantitative measures of effect can also make significant contributions to variance in the estimates of ecological risk. For this risk assessment, the measures are based on the water quality criteria established by the State of Idaho and on habitat suitability indices developed by the U.S. Fish and Wildlife Service. The quantitative water quality criteria have been developed from a wide range of field and laboratory tests and assays. Such tests often show considerable variability in results. Furthermore, they may not always be appropriate for local conditions because of variability in environmental factors and in the response of the target organisms.

In many cases, the exposure period used in the tests or assays used to develop criteria may not be commensurate with exposure periods experienced in the real system. For example, many of the water quality criteria are based on results from bioassays conducted over a period of 48 hours. The results of the bioassays are interpreted in terms of acute or chronic impacts on test organisms. The acute and chronic values are converted to criteria stated in terms of instantaneous or 4-day averages, respectively, in the water body. Interpreting the results of tests and assays in this way is meant to be protective of the ecosystem, but it does introduce uncertainty.

There is even more uncertainty associated with the quantitative interpretation of narrative standards. The narrative criterion for macrophytes is "nuisance" levels. There is no qualitative or quantitative definition of "nuisance." There is general agreement that the levels observed during the period 1990 to 1994 were unacceptable. However, there is as yet no agreement on what quantitative measures of macrophyte growth constitute a violation of these standards. The analysis of macrophyte literature used in this study is limited in scope and may not necessarily reflect local conditions in the study reach. The nuisance criterion should undoubtedly be adjusted for the type of plant community. For example, densities of *P. pectinatus* at a level of 200 g/m^2 might not hinder beneficial uses. Densities of *Cladophora* of this magnitude, because of their surface-floating habit, could severely hinder beneficial uses and therefore be considered a nuisance. Until further discussion and analysis is devoted to this issue, the measurement endpoint for macrophyte biomass should be considered provisional.

9.2.7. Lack of Knowledge

The habitat suitability indices, based on the IFIM procedure used by the U.S. Fish and Wildlife Service (Anglin et al., 1992) in the Swan Falls reach of the Snake River, are valuable as measurement endpoints because they were derived for target organisms found in the study reach. They provide measures of ecological impacts on target organisms attributable primarily to physical stressors. The uncertainty in these indices is due primarily to lack of specific knowledge regarding impacts of the stressors on the target organisms. Some of these habitat suitability indices, for example, are based on surveys of the best professional judgment of regional fisheries biologists. The results of the surveys were converted to indices as part of the IFIM (Anglin et al., 1992) without quantifying the uncertainty in the responses. Even though the results of IFIM modeling are somewhat subjective, they effectively reveal temporal habitat trends.

Because the focus of IFIM is on instream flows, the habitat suitability indices give considerable weight to the physical parameters, i.e., depth and velocity. Some researchers (e.g., Mathur et al., 1985) have reported there is little evidence showing correlation between these habitat indices and fish biomass. Proponents of the methodology (e.g., Armour and Taylor, 1991) acknowledge the lack of post-project monitoring to document the value of IFIM, while noting that IFIM has become widely accepted as a tool for investigating the impacts of flow regulation on aquatic resources. The rationale for adapting the IFIM methodology for ecological risk assessment in the study reach was based primarily on this acceptance. However, given the manner in which the indices were developed and the lack of documented support for the method, the results reported here are likely to have a high degree of uncertainty.

9.3. CONCLUSIONS

The Middle Snake River is clearly altered from the natural lotic ecosystem that existed prior to construction of impoundments and water withdrawal. It is now a series of reservoirs with few remaining rapids. The result is a system dominated by low-flow dynamics. In addition, the springs that should provide cool, clear, low-nutrient water are being preempted for aquaculture facilities.

Loading of nutrients, sediments, and thermal energy into this ecosystem has caused the natural cold-water populations to decline and eutrophication to accelerate. Natural events (high flows) can alleviate this trend. High flows during 1996-1998 scoured much of the deposited sediments and provided higher flows and cooler temperatures for aquatic growth and survival. However, the cycle of eutrophication and loss of diverse habitats will be repeated when the flows are reduced. The only way to sustain a viable cold-water population is to reduce all anthropogenic loadings to negligible levels unless the natural hydrograph can be reinstated.

Sediment dynamics are as important as water column dynamics in providing the necessary habitat for growth and survival of native biota of the river. Sediment deposition alters habitat by providing substrate for the growth of macrophytes and by smothering gravel beds that are normally niches for fish and invertebrates.

The physical removal of both the nutrient-laden substrate and the macrophytes by transport processes was the primary factor leading to reduction of macrophyte biomass in the study reach (Clark, 1997b). The methods described here did not include an analysis of the physical processes characterizing sediment deposition and scouring. Additional studies are necessary to describe these processes and their effects on macrophyte and epiphyte mortality.

Eutrophication leads to several ecological changes in the structure and function of freshwater ecosystems. Such changes include increased plant (macrophyte) production and shifts in species composition, decreased water transparency, and shifts in the consumer community towards undesirable species. The results of eutrophication are manifested in a macroinvertebrate community comprised predominantly of pollution-tolerant taxa. The restriction of the community to pollution-tolerant taxa resulted in the increased abundance of the species present, particularly in the early 1990s. Additionally, the habitat conditions of the Middle Snake River in the early 1990s favored the distributional expansion and increased abundance of the exotic snail *Potamopyrgus antipodarum.*

The impoundment and subsequent alteration of hydrologic regimes is the most disruptive of all common anthropogenic forces acting on rivers (Ligon et al., 1995; Sedell et al., 1990). In the Middle Snake River, impoundment has altered the hydrologic regime, created lentic environments, and reduced water velocities in many of the remaining lotic reaches. These changes have resulted in warmer water temperatures, increased deposition of fine sediments, increased residence times for dissolved nutrients, and the proliferation of aquatic macrophytes (Royer et al., 1995; Thomas et al., 1995; Falter and Carlson, 1994). The aquatic macroinvertebrate community of the Middle Snake River now reflects the altered riverine conditions. The above changes in the river have created a habitat favoring warm-water, sediment-dwelling, and generally pollution-tolerant taxa.

Salmon and steelhead trout once spawned in the Snake River as far upstream as Auger Falls, indicating that the Middle Snake River was a clear, cobble/gravel-bottomed, cold-water river (Evermann, 1896). Indeed, Evermann (1896) stated, “The spawning grounds of chinook salmon in [the] Snake River between Huntington and Auger Falls have been ... the most important in Idaho.” Although anecdotal, descriptions such as Evermann’s suggest the Middle Snake River once displayed characteristics typical of rivers productive for anadromous salmon. A return of a part of this habitat is needed to sustain cold-water resources in this area.

10. MANAGEMENT IMPLICATIONS OF THE MIDDLE SNAKE RIVER RISK ANALYSIS

The assessment process, in particular the problem formulation, furthered a better understanding of environmental problems. Developing the conceptual model provided participants with an improved understanding of the interrelationships between various components of the ecosystem and how human activities contribute to environmental problems within the watershed. Assessment endpoint selection helped ensure data that was collected and analyzed would be relevant to decision-making needs. The risk assessment helped lend further credence to what many professional resource manages had long conjectured about problems within the watershed, thereby providing more scientific support for actions to address problems. For example, there is now a better understanding of the contribution of sediment to the changes in ecosystem, including eutrophication and decline of fish species. This information is useful to managers as they begin to regulate the sediment sources.

The conceptual model and analysis plan were helpful to the State of Idaho Department of Environmental Quality (IDEQ) in designing monitoring plans for the river. The simulation techniques as described in appendix D, were the IDEQ to prepare its Nutrient Management Plan (1995) for phosphorus in the river. The mathematical models predicted phosphorus levels in the river and these were compared with macrophyte growth rates to develop the phosphorus Total Maximum Daily Load (TMDL) concentration. The qualitative analysis of literature and field surveys of the survival threshold limits of freshwater biota was used by EPA in its review of the TMDL for sediment, which was completed by the IDEQ in 1999. The qualitative analysis also provided information for the U.S. Fish and Wildlife Service for its Snail Recovery Plan (1995). Assessment findings provided information for the National Pollution Discharge Elimination System (NPDES) (1999) permits for aquaculture facilities on the Middle Snake River.

In 1992, the simulation techniques developed as part of the risk analysis as well as the conceptual model were used by EPA in their determination of the impacts from a proposed new hydroelectric facility (Auger Falls) on the Middle Snake River. The Federal Energy Regulatory Commission (FERC) was convinced by EPA's risk analysis that cumulative impacts should be addressed when reviewing new or continued licensing for hydroelectric facilities. The EPA risk analysis also provided the community groups with a more robust analysis of the possible impacts of impoundments on the river.

In 1995, a Watershed Advisory Group was formed from the State of Idaho Nutrient Management Plan committee. The state legislation established such groups to provide guidance to the state on pollution control efforts that would lead to full support of designated uses for high-priority water bodies as defined in the state water quality standards. The legislation specified that members of the watershed advisory groups should include representatives of local governments and agencies. The comprehensive analysis of the Middle Snake watershed and the compilation of information from literature and field surveys will be useful to the Watershed Advisory Group as it begins comprehensive strategic planning for watershed protection.

The simulation methods developed in this watershed can be used to test management scenarios. For instance, how does changing river flow, sediment input, or point and nonpoint sediment dynamics impact different macrophyte biomass levels or fish populations. This allows all those interested in land-use activities, as well as resource protection, to explore such alternative management options for the river and to make more informed decisions. The simulation techniques developed through this risk analysis can also be applied to other watersheds and similar conditions. It is currently being used on the main stem of the Columbia and Snake rivers for examining alternative impoundment options to protect salmonid habitat.

The biggest benefit from performing this watershed ecological risk assessment is that a number of federal, state, and local environmental agencies and academics, interested citizens, and industrial groups came together to share data, explore, and develop solutions and undertake actions within the watershed.

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APPENDIX A. PARTICIPANTS IN THE PROTECTION OF THE MIDDLE SNAKE RIVER

Federal

U.S. Environmental Protection Agency (U.S. EPA)
Department of the Interior
U.S. Fish and Wildlife Service (USFWS)
Bureau of Land Management/Minerals Management Service
National Biological Survey
Department of Energy
Federal Electric Regulatory Commission
Department of Agriculture
National Park Service
U.S. Geologic Survey
Northwest Power Planning Council

State

Idaho Department of Health and Welfare
Division of Environmental Quality
Idaho Department of Water Resources
Idaho Department of Fish and Game
Idaho Fish and Game Commission
Idaho Water Board
Idaho Department of Parks and Recreation

County/Local

Mid Snake River Planning Group

Private Organizations

Idaho Power Company
North Side Canal Company
The Nature Conservancy

Natural Heritage Program

The Research Community

The University of Idaho

Idaho State University

University of California at Irvine

APPENDIX B. ECOLOGICAL COMPONENTS OF THE MIDDLE SNAKE RIVER ECOSYSTEM

Wetland and shoreline plants found in the Snake River (from Stanford, 1942; Dey and Minshall, 1992). Dey and Minshall species are marked with an asterisk. The names are in accordance with Hitchcock and Cronquist (1981).

Scientific Name	Common Name
Salix lasiandra	Willow
Populus trichocarpa	Cottonwood
Nepeta cataria	Catnip
Solanum triflorum	Nightshade
Veronica americana	American brooklime
Solidago missouriensis	Goldenrod
Rumex persicarioides	Dock
Vicia americana	Vetch
Glycyrrhiza lepidota	Licorice
Apocynum cannabinum	Dogbane
Verbena hastata	Verbena
Mentha arvensis lanta	Mint
Helenium autumnale	Sneezeweed
Xanthium pensylvanicum	Cocklebar
Bidens cernua	Beggar-ticks
Artemisia sp.	Mugwort
Sarcobatus sp.	Greasewood
Phragmites communis	Reed
Paspalum distichum	Knotgrass
Polypogon monspeliensis	Beard-grass
Cyperus strigosus	Flatsedge
Eleocharis palustris	Spike-rush
Scirpus validus	Soft-stem bulrush
Typha latifolia	Cat-tail
Polygonum natans	Doorweed

Scientific Name	Common Name
Polygonum lapathifolium	Doorweed
Sagittaria sp.	Arrowhead
Potamogeton epihydrus	Pondweed
Potamegeton pectinatus	Pondweed
Ceratophyllum demersum	Hornwort
Rorippa nasturtium	Cress
Lemna minor	Duckweed
Azolla sp.	Water-fern
Toxicodendron diversiloba	Sumac
*Potamogeton crispus**	Pondweed
*Potamogeton foliosus**	Pondweed
*Elodea nuttallii**	Waterweed
*Elodea canadensis**	Waterweed
*Ranunculus spp.**	Buttercup
*Myriophyllum spicatum**	Water-milfoil

Fish species found in Middle Snake River between King Hill and Milner Dam (personal communication with Idaho Department of Fish and Game, 1993; Idaho Division of Environmental Quality, 1995; Maret, 1995).

Scientific Name	Common Name
Family: Acipenseridae - Sturgeons	
Acipenser transmontanus [1,2,3,4]	White sturgeon
Family: Salmonidae - Trouts	
Oncorhynchus clarki [1]	Cutthroat trout
Oncorhynchus mykiss [1,4]	Rainbow trout
Oncorhynchus mykiss gairdneri [5]	Redband trout
Prosopium williamsoni [1,4]	Mountain whitefish
Salmo trutta [4,6]	Brown trout
Family: Cyprinidae - Carps and Minnows	
Cyprinus carpio [4,6]	Common carp
Ptychocheilus oregonensis [4]	Northern pikeminnow

Mylocheilus caurinus [4]	Peamouth
Acrocheilus alutaceus [4]	Chiselmouth
Richardsonius balteatus [4]	Redside shiner
Rhinichthys osculus [4]	Speckled dace
Gila atraria [6]	Utah chub
Rhinichthys cataractae [4]	Longnose dace
Rhinichthys falcatus [4]	Leopard dace
Family: Catostomidae - Suckers	
Catostomus columbianus [4]	Bridgelip sucker
Catostomus macrocheilus [4]	Largescale sucker
Catostomus platyrhynchus [4]	Mountain sucker
Catostomus ardens [7]	Utah sucker
Family: Ictaluridae - Bullhead catfish	
Ictalurus punctatus [1,2,4,6]	Channel catfish
Ameiurus nebulosus [4,6]	Brown bullhead
Ameiurus melas [4,6]	Black bullhead
Family: Centrarchidae - Sunfishes	
Micropterus dolomieu [1,2,4,6]	Smallmouth bass
Micropterus salmoides [1,4,6]	Largemouth bass
Lepomis gibbosus [4,6]	Pumpkinseed
Pomoxis nigromaculatus [4,6]	Black crappie
Lepomis macrochirus [4,6]	Bluegill
Family: Percidae - Perches	
Perca flavescens [1,4,6]	Yellow perch
Stizostedion vitreum [4,6]	Walleye
Family: Cottidae - Sculpins	
Cottus bairdi [4]	Mottled sculpin
Cottus greenei [3,4]	Shoshone sculpin
Cottus beldingi [4]	Paiute sculpin
Cottus confusus [4]	Shorthead sculpin
Cottus rhotheus [4]	Torrent sculpin
Family: Sciaenidae - Drums	
Aplodinotus grunniens [4,6]	Freshwater drum

Native fish species extirpated from the Middle Snake River

Onchorhynchus tshawytscha	Chinook salmon
O. *kisutch*	Coho salmon (possible inhabitant)
O. mykiss	Steelhead trout
Lampetra tridentata	Pacific lamprey

Notes for fish species:

[1] Game fish in the Middle Snake River (IDEQ, 1995).

[2] Spawning fish (IDEQ, 1995).

[3] Considered a Species of Special Concern by the State of Idaho.

[4] Fish fauna of the Snake River drainage below Shoshone Falls (Bowler et al., 1992; Bowler, personal communication, 1992).

[5] The only pure surviving population of redband trout is believed to be in King Hill Creek; hybrids are found in other tributaries.

[6] Non-native species. Five additional non-native species likely present are:

Tilapia mossambica	Mozambique tilapia
T. zelli	Redbelly tilapia
T. nilotica	Nile tilapia
Lepomis cyanellus	Green sunfish
L. microlophus	Redear sunfish

[7] Federal Energy Regulatory Commission (1990).

Some **Molluscans** found in the Middle Snake River (from Frest and Bowler, 1992; Dey and Minshall, 1992).

Scientific Name	**Common Name**
Class Gastrioida (Snails)	
Ancylidae	
Ferrissia parallelus	
Ferrissia rivularis	
Hydrobiidae	
Fluminicola columbiana [1]	Columbia River spire snail
Fluminicola hindsi	
Potamopyrgus antipodarum	
Pyrgulopisis idahoensis [2]	Idaho springsnail
Taylorchoncha serpenticola [3]	Bliss Rapids snail
Potamopyrgus antipodarum [4]	New Zealand mudsnail
Lancidae	
Fisherola nuttalli [1]	Giant Columbia River limpet
Lanx sp. [2]	Banbury Springs lanx (undescribed)
Fossaria bulimoides	
Fossaria dalli	
Fossaria exigua	
Fossaria modicella	
Fossaria parva	
Fossaria obrussa	
Stagnicola caperata	
Stagnicola catascopium	
Stagnicola hinkleyi	
Radix auricularia [4]	
Physidae	
Physa natricina [2]	Snake River physa
Physa mexicana	
Physella gyrina	
Physella integra	

Scientific Name	**Common Name**
Planorbidae	
Gyraulus parvus	
Planorbella subcrenatum	
Promenetus exacuous	
Vorticifex effusus	
Valvatidae	
Valvata utahensis [2]	Utah valvata snail
Valvata humeralis	
Class Pelecypoda (Clams)	
Corbiculidae	
Corbicula fluminea [4]	Asian clam
Margaritiferidae	
Margaritifera falcata [5]	
Sphaeriidae	
Musculium lacustre	
Musculium securis	
Pisidium compressum	
Pisidium caesertanum	
Pisidium insigne	
Pisidium nitidum	
Pisidium pauperculum	
Pisidium punctatum	
Pisipium variabile	
Sphaerium nitidum	
Sphaerium patella	
Sphaerium striatinum	
Unionidae	
Anodonta californiensis [1]	California floater
Gonidea angulata	

Notes for molluscan species:

[1] Species of concern. Conclusive data not currently available for listing under the Endangered Species Act.

[2] Listed as an endangered species under the Endangered Species Act.

[3] Listed as a threatened species under the Endangered Species Act.

[4] Non-native species.

[5] Extirpated from the Middle Snake River.

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APPENDIX C. LIFE HISTORIES OF DOMINANT MACROPHYTE SPECIES IN THE MIDDLE SNAKE RIVER

Potamogeton pectinatus

P. pectinatus is a leathery-stemmed, perennial angiosperm. Stems may be up to 2 m long with 3- to 15-cm leaves about 1 mm wide. All stems and leaves are submerged and tough and leathery. *P. pectinatus* does well in high-velocity waters, well over 1 m/sec. The fruit, 3 to 4 mm long and short-beaked, is prime waterfowl food, especially for dabbling ducks. Perennial rhizomes are much-branched and tipped by fleshy tubers. *P. pectinatus* prefers eutrophic (Grasmuck et al., 1995), alkaline waters, and hence is probably the most common rooted macrophyte across waters of the arid and irrigated Western United States (Falter et al., 1974). Biomass levels of 500 to 1,000 g/m^2 are not rare in irrigation drain waters. In the inland Northwest, by the first week of May, shoots began to emerge from perennial rhizomes when the water temperature exceeds 4°C to 5°C. Stem growth develops slowly until mid-June, when water temperatures and light energy increase towards summer maxima. While growth of foliage slows in July, flowering begins then with fruit starting to form by mid-July. The fruits mature by early August and begin to fall to the sediments. In early September new growth occurs in the leaf axils of the shoot. These growths develop into leaflike branches or axillary tubers which are well developed by October. Much of the foliage dies back by mid-October.

Reproduction in *P. pectinatus* is either by sexual or vegetative means. The monoecious (single-sex) flowers are fertilized with help of movement of pollen along the surface of water to the stigmas. This results in the formation of a new plant embryo or seed/fruit. Vegetative propagation occurs by stolons branching out at the base of the main shoot. At each node on the stolon a new set of roots and shoot may develop, potentially creating a new plant.

Potamogeton crispus

P. crispus was introduced into North America in the mid-nineteenth century from Eurasian waters and can now be found in all 48 contiguous States. It is a submerged aquatic vascular plant that perennates by detaching a fleshy winter bud. Vegetative reproduction is very important to this species because sexual reproduction has not yet been observed. Flowering occurs in the early spring but formation of seeds is little understood. *P. crispus* is a cool-water strategist that can overwinter as the entire green plant under ice or via winter buds. The thick, crinkled, crisp leaves grow very quickly in spring as temperature and light intensity increase. This allows *P. crispus* to establish itself early in the growing season. *P. crispus* has three vegetative forms: (1) the previously mentioned winter bud, which can survive low-light conditions under ice; (2) the spring form stimulated by warming water, which is the plant form one sees from mid-spring to June and early July; and (3) the dormant apices (viable bud tips, a vegetative reproduction mechanism). The dormant apices detach before germinating in autumn, then begin to grow into a flat, narrow-leaved winter form. Spring germination of dormant apices is controlled by temperature (Nichols and Shaw, 1986; Stuckey et al., 1978). The magnitude of vegetative propagation can be phenomenal, especially in warmer waters. Despite being a cool-water strategist in the winter, this plant's optimal temperature for photosynthesis is about 30°C. Yeo (1966) planted a single dormant apex of *P. crispus* in a 6 m^2 container and counted 23,250 dormant apices produced by the end of the growing season. Dormant apex densities of 23 to 1,650/m^2 have been found in natural *P. crispus* populations (Kunii, 1982).

P. crispus prefers moderately to very alkaline waters and does very well in eutrophic and even saline waters. Like *E. canadensis*, the plant is considered an indicator of eutrophic conditions (Hellquist, 1980; Nichols and Shaw, 1986). It similarly has a high tolerance for pollution. Under these borderline conditions for most native species, *P. crispus* may often dominate a plant community, at least early and mid-growing season. Its habit for summer die-back exacerbates late-summer planktonic algae blooms by sudden release of nutrients pumped from sediments earlier in the season. Another reason for the plant's explosive growth potential is undoubtedly its ability to absorb nutrients either from sediments via roots or from the water column via shoots and leaves (Carignan and Kalff, 1980; Nicholls and Shaw, 1986). Like milfoil and *E. canadensis*, *P. crispus* may also experience catastrophic die-off where it may nearly disappear from a habitat where it previously dominated. All three species, however, are remarkably disease-free.

Several researchers have demonstrated the ability of *P. crispus* to thrive in highly turbid waters or deep in the water column at low light intensity. It is considered a deepwater species, one that typically develops at depths where light intensity is $\leq$15% of full sunlight (Nicholls and Shaw, 1986).

Ceratophyllum demersus

C. demersus is a submerged, free-floating, rootless aquatic vascular. This species usually overwinters by surviving in densely crowded, dormant stem apices and sometimes as an intact vegetative plant body, especially in deeper water covered by ice (Stuckey et al., 1978). *C. demersus* reproduces sexually and by fragmentation (Tarver et al., 1979). In sexual reproduction the pollen is most often released from monoecious flowers. Pollen is released from the stamen at some time in the middle of summer and sinks through the water until caught on a stigma. This results in the formation of fruit, which needs warmer temperatures to mature. The fruit is a persistent, hooked nutlet. The seed germinates at the bottom of an enclosing water body. When plantlets reach about 7.5 cm (3 inches) in length, they can rise to the surface to float. These young shoots form in the spring. *C. demersus* is more sensitive to colder temperatures than are other aquatic plants (Arber, 1920; Mason, 1969). The plant seems to prefer slow-moving streams, quiet ponds, and protected bays on lakes (Tarver et al., 1979).

C. demersus is clearly favored by polluted, enriched waters. Kurimo (1970) found the plant to prefer locations below industrial and urban waste inputs, so long as the water was not too acid. In Japanese waters impacted by irrigation, Kunii (1991) used canonical analyses to place macrophytes on environmental gradients. *C. demersus* occurrence was associated with higher pH, conductivity, alkalinity, Ca, Mg, and Na.

Elodea canadensis

E. canadensis is a submerged, branching, leafy, perennial but poorly rooted aquatic vascular. Native to North America from Quebec to the Gulf Coast States and west to the Pacific (although notably absent from the higher Rockies), it often behaves as an exotic in its rapid spread and dense, luxuriant growths. It reproduces almost entirely vegetatively, but will rarely reproduce sexually. The sexual fecundity of this species is low and depends on the dioecious form of the plant. The staminate plants are rather rare compared to their pistillate counterparts (Arber, 1920). Vegetative reproduction is therefore important to *E. canadensis* as seed formation is rare. Rapid propagation can result when a branch containing one or more nodes is broken off and continues to grow. Such stem fragmentation is the most common method of *E. canadensis* reproduction. *E. canadensis* can withstand cold temperatures in the winter by living in dense colonies. It has commonly been observed that the foliage can remain green throughout the winter under ice. This species is a cold-water strategist and thus has an edge in the spring when temperatures and light intensities increase and its foliage is already in place and ready to grow. Unlike *P. crispus*, the growing season for *E. canadensis* extends throughout early spring to late fall.

E. canadensis may obtain nutrients either from soil or water. A number of studies have shown both occurring, depending upon relative nutrient concentration of the water or sediments. Opportunism seems to be the rule (Nicholls and Shaw, 1986; Stuckey et al., 1978). Its depth preference is broad, possibly dominating communities from 1 to 12 m depth.

Hydrodictyon

Hydrodictyon (water net) is a filamentous green alga in the class Chlorophyceae. This alga is colonial, forming large cylindrical nets up to 60 cm long. Individual cells are also cylindrical and can themselves be up to 1 cm long. At each end, they are attached to three other cells, the result producing a reticulate net of pentagonal or hexagonal shapes. Colonies resemble a nylon mesh stocking in the current. In the Middle Snake River, these tubes are aligned parallel to the current with the tubal opening facing the current. The tubes are very efficient detrital traps, eventually settling to the stream bottom where they add to the sediment store of organic matter, energy, and nutrients. Water net occurs worldwide in slowly flowing or nearly still freshwaters, where it can form large growths, particularly in nutrient-rich waters. This species prefers eutrophic waters. In the Middle Snake River, *Hydrodictyon* grow in the top half of the water column in dense rooted macrophyte beds where there is just enough current to fill out the nets. This genus has the distinction of being one of the earliest known alga mentioned in literature, being referenced in one ancient Chinese poem, *Bible of Poems*.

Reproduction is quite complex. Asexual reproduction occurs through the formation of autocolonies in which the cytoplasm is forced to the outer surfaces of the cell wall by a large vacuole. The cytoplasm then divides to many protoplasts, small flagellated zoids. Eventually, there may be up to 20,000 flagellated cells linked together in the form of a haplontic (adult cells are 1N) netlike colony.   Asexual reproduction continues when these same zoospores may disperse to form individual water nets. A net with 10,000 cells could theoretically produce 10,000 nets in the water. Such efficient reproduction explains why in a very short time, *Hydrodictyon* may form thick and tangled mats in streams and irrigation canals.   Judging from the appearance of detritus-laden *Hydrodictyon* nets in the Mid-Snake River, the colonies apparently trap and remove drifting sestonic nutrients and organic matter from the water column. Given its rapid reproductive rate and net structure, *Hydrodictyon* undoubtedly has a significant role in nutrient and organic matter removal from water flows through the aquatic macrophyte beds.

Cladophora

Cladophora, another filamentous branched green alga, is sometimes called “blanket weed.” Its life form is macrophytic, branched filaments that initially develop attached to rocks via a holdfast or rhizoidal outgrowths or even as epiphytic growths on vascular macrophytes. In streams, attached streamers may attain lengths of 10 m. Older colonies often leave the substrate and become self-supporting in the water column. If not crowded, the growth form may develop as bushy tufts waving in the current. In streams, the clusters may form streaming hanks of algae tissue. The plant may take advantage of vascular macrophyte growths on the sediments and a rich water column environment by forming blanketing mats on the vasculars. Light restriction to underlying macrophyte communities may be extreme and responsible for vascular macrophyte declines (Phillips et al., 1978; MacMillan, 1992).

Cladophora may be marine or freshwater, but does best in areas of some water movement, such as streams or tidal flows. The genus is one of the better plant indicators of organic loading and nutrient enrichment. Pitcairn and Hawkes (1973) found high correlation between *Cladophora* growth and soluble inorganic phosphorus concentration up to about 2,000 μg/L. It is a very common form below sewage outfalls, aquaculture facilities, and food processing facilities, and in sublittoral zones of bathing beaches. Its presence nearly always indicates enhanced concentrations of soluble organics in the water. *Cladophora* is known for its very high biomass levels, sometimes exceeding 1,000 g/m^2 dry weight.

Most reproduction is by asexual cell fission, resulting in filaments several decimeters long. In the freshwater species, there is a cytological alteration of generations, the plantlike phase being the sporopyte. In the winter months the sporophytes produce diploid planogonidia or aplanogonidia depending on the species. During the spring the sporophyte forms sporangia where meiosis occurs. This results in the formation of haploid nuclei representing the spores. The spores germinate within the sporangia. This leads to the production of biciliate gametes, which are then released to join via syngamy. Following syngamy, a 2N zygote is formed and the zygote attaches to some substrate growing into a new sporophyte, which grows into a vegetative filament.

Spirogyra

Spirogyra is a genus of filamentous green algae sometimes referred to as water silk. Filaments can be several decimeters long, are unbranched, and do not require attachment to substrate. *Spirogyra*s prefer a range of living conditions that is rather wide, from deep, cold springs to shallow, warm ponds, or from very hard water areas to waters with large amounts of organic acids. Cooler conditions are preferred because they are especially common in spring and early summer. They are ubiquitous in habitat selection and therefore, are not good biotic indicators of any particular range of water productivity. Generation time of a colony is variable, but 2 months is a common time period from zygote germination to filament senescence. Vegetative growth peaks at from 2 to 6 weeks, followed by sexual reproduction as well as formation of and burial of zygotes (fertilized 2N embryos) in the substrate. These zygotes are quite resistant to anaerobic conditions in decomposing muds over fall and winter.

Most growth is vegetative through intercalary cell division splitting at the cross walls. Sexual reproduction is common and complex. Starting with a *Spirogyra* zygote, which can be thought of as the sporophyte, meiosis will occur when conditions are favorable. The zygote is a very resistant cell able to withstand extreme cold or desiccation. It is able to do this by being released after conjugation and settling into the mud, covered with slime. So, when meiosis occurs in the diploid zygote, four haploid nuclei are produced and only one nucleus will remain after the other three break down and disappear. When the remaining haploid cell begins to divide, it will form a young gametophyte. The young gametophyte is either a female or male filament. These are the 1N filaments commonly seen. *Spirogyra* is therefore haplontic, where 2N exists only in the zygote and “adult” filaments, either male or female. Each gametophyte cell contains one haploid gamete. Conjugation, the mode of sexual reproduction, occurs when the male and female filaments lie appressed and the male gamete passes into the female gametangium by a conjugation tube. The male and female gametes fuse by syngamy, which results in a new diploid zygote. The gametophyte filaments can also reproduce by vegetative means when a segment breaks off.

Enteromorpha

Entermorpha spp. is a filamentous green alga genus that grows as hollow, intestinal-shaped tubes up to 15 cm or more in length. The genus may be freshwater or marine; freshwater forms are considered halophytic, doing well in brackish water. It is a common alga around the world in tropical tidal pools. Freshwater species do best in alkaline waters (about 300 to 500 mg/L total alkalinity) and in high phosphorus concentrations, and can tolerate high ammonia. The entrail-shaped sporophyte is usually attached to some type of substrate. Species of this genus have a typical algal alteration-of-generations life cycle. The 2N sporophyte produces sporangia where meiosis occurs resulting in haploid (N) motile spore formation. The spores are released through an opening in the outer cell wall of the sporophyte cells to the surrounding water. The spores move away, eventually germinating and attaching to some substrate. A gametophyte phase then develops where the gametophyte is tubular and filamentous, resembling the sporophyte. There are male and female gametophytes which produce and release male and female haploid, motile gametes. A male and female gamete fuse by syngamy and produce a diploid zygote, which germinates and forms a new sporophyte.

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APPENDIX D

Analysis of Ecological Risk in the Mid-Snake River Using Simulation Methods

March 10, 2000

John R. Yearsley
U.S. Environmental Protection Agency
Reagion 10
Seattle, WA

C. Michael Falter
University of Idaho
Moscow, ID

APPENDIX D. ANALYSIS OF ECOLOGICAL RISK IN THE MID-SNAKE RIVER USING SIMULATION METHODS

D.1. INTRODUCTION

Mathematical models of mass and energy flow in an aquatic ecosystem are used in the assessment of the Study Reach, the Middle Snake River from Milner Dam (river mile [RM] 640.0) to King Hill (RM 545.3) (Figure D-1), to obtain quantitative estimates of ecological risk. The mathematical models simulate physical, chemical, and biological state variables that characterize important features of the aquatic ecosystem. The probabilistic measures developed from the simulations will be compared with the relevant measurement endpoints to estimate exposure and effect in the Snake River. The probabilistic aspects of the analysis arise from the uncertainty and variability in environmental factors such as hydrology and meteorology and in the loadings, both anthropogenic and natural, of nutrients, toxic substances, and thermal energy.

The mathematical model is based on a set of hypotheses regarding physical, chemical, and biological processes in the aquatic ecosystem. These hypotheses are derived from laboratory and field experiments and from conservation laws for mass, energy, and momentum. State variables simulated by the mathematical model are compared with field observations from the Study Reach to estimate model parameters and to characterize the uncertainty of the simulated results. The mathematical model is then used to make quantitative estimates of ecological risk for certain components of the aquatic ecosystem.

D.2. MODEL DEVELOPMENT

The process of model development includes (1) stating of hypotheses that form the basis for the model; (2) estimating the parameters that control the flow of mass and energy in the model using laboratory and field studies and other available research; and (3) characterizing the uncertainty of simulated results from the mathematical model. The hypotheses forming the basis for the mathematical model of the Study Reach are:

- Major features of the Study Reach can be described in terms of compartments between which there can be flows of energy, material, and information.

- The flows of energy, mass, and information between ecosystem compartments can be described mathematically on the basis of the results of existing ecosystem theory.

- Probabilistic models can be developed for describing the uncertainty and variability of environmental driving forces including meteorology and hydrology.

- Probabilistic models can be developed for describing the uncertainty and variability of loadings of nutrients, toxic substances, and thermal energy from both anthropogenic and natural sources.

- Measurement endpoints for the risk analysis can be derived from the State of Idaho's water quality standards and from habitat suitability measures developed from biological research.

D.2.1. Dynamic Model of Mass and Energy Balances

In Figure D-2, ecosystem driving forces, characterized by variability found in the Study Reach, are input to a dynamic model of mass and energy balance. The simulated state variables output from the model will have variability derived from the input driving forces. The model output can be used to characterize stressors and exposure where simulated state variables correspond to measures of ecosystem effects. To implement the dynamic model of mass and energy, the Study Reach has been divided into two major ecosystem components: one that describes the chemical, physical, and biological characteristics of the moving water column and one that describes the benthic plant community attached to or associated with the river bottom. The flow of energy, mass and information for these components is shown in Figures D-3 and D-4.

D.2.1.1. *Water Column*

For purposes of capturing spatial variation, the water column is segmented as well-mixed compartments organized either longitudinally as in Figure D-5 or vertically as in Figure D-6. The longitudinal organization is used to describe freely flowing segments or run-of-the-river reservoir segments. The vertical organization is used to describe segments where vertical stratification of water quality constituents may be important. Within each of the well-mixed compartments, the flow of energy and material is the same as shown in Figure D-3.

The mathematical description of mass and energy flow in the water column described in this report is based on the mathematical model RBM10 (Yearsley, 1991). RBM10 has been used as a decision support tool in a number of river basins in the Pacific Northwest, including the Snake River above Milner Dam (Yearsley, 1976) and the Spokane River (Yearsley and Duncan, 1988). RBM10 makes use of concepts that have been used in other modeling efforts (e.g., Thomann et al., 1975; Patten et al., 1975; DiToro et al., 1975; Chen and Orlob, 1975; Scavia, 1980) and is conceptually similar to these models. State variables in the water column simulated by this model are given in Table D-1.

The general assumptions associated with the mathematical development of the model for water column state variables are:

- State variables of the ecosystem can be characterized by volume-averages over a given computational element (Figure D-5 for free-flowing rivers or run-of-the-river reservoirs and Figure D-6 for stratified reservoirs) and a finite time interval.

- Horizontal and vertical advection and vertical eddy diffusion are the primary physical processes for water and mass transport.

- The vertical coefficient of eddy diffusivity is the same for all state variables.

- Rate constants for the various reactions do not change over a given length segment.

- The river system can be divided into a finite number of segments within which cross-sectionally averaged hydrodynamic characteristics are constant.

- Hydrodynamic characteristics of free-flowing river segments and run-of-the-river reservoirs can be expressed as a simple function of the flow in any segment.

- The river flow varies gradually such that hydraulic characteristics can be estimated using the standard methods for calculating steady, gradually varied flow in natural or man-made channels.

- Hydrodynamic characteristics of stratified reservoir segments are a function of the density structure of the reservoir. The density structure in freshwater systems is a function of water temperature and the concentration of suspended material.

- The time required for flow in a reach to adjust to changes in elevation is small compared with the travel time of some constituent.

Given these assumptions for a state variable, C, representing a water column constituent that is spatially averaged over a computational element, the general conservation equation in the i^{th} free-flowing river or run-of-the-river reservoir segment can be described by the mass balance equation:

$$\frac{d(C_i V)_j}{dt} = \Delta(QC_i)_x + \sum_{n=1}^{N_s} (Q_p C_{ip})_n + \Phi_{ij} - \Gamma_{ij} \qquad \text{(D-1a)}$$

Similarly, for the jth stratified reservoir segment, the general conservation equation is

$$\frac{d(CV)ij}{dt} = \Delta(QC)_x + \Delta(QC)z + \Delta(KAC)_z + \sum_{n=1}^{N_s} (Q_p C_p)_n + \Phi_{ij} - \Gamma_{ij} \qquad \text{(D-1b)}$$

where

V_j = the volume of the jth water column computational element, where j refers to the segment number;

C_i = the length- and time-averaged value of the ith water column state variable over the jth computational element;

$\Delta(QC)_x$ = the advective transfer in the longitudinal (x-) direction;

$\Delta(QC)_z$ = the advective transfer in the vertical (z-) direction;

$\Delta(QC)_n$ = the transfer of flows from the computational element due to inputs or outputs such as point source discharges, nonpoint source return flows, and withdrawals for drinking water or irrigation;

$\Delta(KAC)_z$= the eddy diffusion in the vertical (z-) direction;

A_{ij} = the surface area of the ijth element;

Φ_{ij} = the source term for the state variable, C_i, in the jth element; and

Γ_{ij} = the sink term for the state variable, C_i, in the j element.

For all state variables, gains and losses due to the physical processes of advection and eddy diffusion are treated in the same manner. The source, Φ_{ij}, and sink, Γ_{ij}, for each of the state variables are determined from existing knowledge of physical, chemical, and biological processes.

The rates at which mass and thermal energy are transported, or advected, $\Delta(QC)_x$ and $\Delta(QC)_z$, are determined by the hydrodynamics of the river system. This implementation of the model assumes the river flow varies gradually and that the hydraulic characteristics can be estimated using the methods for calculating steady, gradually varied flow developed by the U.S. Army Corps of Engineers' Hydrologic Engineering Center (HEC). The model used for the hydraulic analysis was HEC-RAS (HEC, 1995). This model cannot be used to analyze the effects of storm events or high-frequency flow fluctuations associated with load-following at hydroelectric facilities. However, the model is relatively easy to implement and provides sufficient flexibility to evaluate the effect of seasonal and long-term changes in river flow.

The hydraulic characteristics of the river system are specified in the ecosystem model in terms of simple relationships between depth and flow and depth and velocity. That is,

$$D = A_d \; Q^{B_d} \tag{D-2}$$

where

D = the average depth of the river segment, feet;
Q = the river flow, cfs;
A_d = a coefficient determined from hydraulic data;
B_d = a coefficient determined from hydraulic data;

and

$$U = A_u \; Q^{B_u} \tag{D-3}$$

where

A_u = a coefficient determined from hydraulic data; and
B_u = a coefficient determined from hydraulic data.

The coefficients A_d, B_d, A_u, and B_u are determined using the steady, gradually varied flow, or step-backwater model, HEC-RAS (HEC, 1995). Given data describing cross-sectional elevation profiles, riverbed elevation, and friction factors as a function of distance along the river, HEC-RAS provides estimates of river surface profiles at various flows. These estimates are obtained by balancing the energy associated with friction losses, changes in riverbed elevation, and changes in river velocity. The output from the models includes estimates of river depth and river velocity at these flows. Regression analysis is performed on this output to estimate the coefficients in Equations D-2 and D-3.

Diffusion-like processes, $\Delta(KAC)_z$, characterize the vertical transport of mass and thermal energy by random motions, or turbulence, in the reservoir segments of the river. The coefficient of eddy diffusivity, K_z, can be estimated from available data. For example, methods for estimating the coefficient of eddy diffusivity for water temperature are described by Water Resources Engineers (1968). Generally, it is assumed the coefficient of eddy diffusivity, K_z, is the same for all state variables. When sufficient data are not available for estimating the coefficient of eddy diffusivity, as is the case for the reservoirs in the Snake River, turbulence closure methods such as those described by Bowie et al. (1985) must be used.

An additional assumption is that horizontal transport of mass and thermal energy by random motions, or turbulence, in the river segments is negligible compared to advection processes.

The kinetics of mass and energy flow for the water column state variables simulated by the model (Table D-1) are similar to the kinetics described in other modeling efforts. Bowie et al. (1985) give a comprehensive discussion of the kinetics of models of this type. Process diagrams and the kinetic formulations for mass and energy flow of water column state variables are described in detail in Appendix A.

D.2.1.2. *Sediments*

The sediments associated with the benthic plants in the Study Reach are segmented into well-mixed compartments organized longitudinally. In general (e.g., Ambrose et al., 1993), many of the physical, chemical, and biological processes in the sediments are similar conceptually to those in the water column. However, in this application of simulation methods to risk analysis, the model hypothesis includes sediments only to the extent they provide substrate for benthic plants, including vascular macrophytes, epiphytes, and periphyton.

The flow of mass and energy within the sediments is generally not included in this model. However, the flow of mass and energy within the water column, as it affects uptake by the roots of vascular macrophytes, is included in the conceptual model. Where flow of mass or energy from the sediments are part of the conceptual model, as in the case of nutrient flow to the roots of vascular macrophytes, it is assumed to be unlimited by plant uptake. Similarly, the flow of solids to and from the sediments is assumed to be at steady state. That is, there is neither gain nor loss of substrate due to deposition and scouring.

For benthic component state variables, B_i, the model hypothesis for the flow of mass and energy is:

$$\frac{d(BV)_{ij}}{dt} = \Phi_{ij} - \Gamma_{ij}$$

where

B_i = the length- and time-averaged value of the i^{th} benthic component state variable over the j^{th} computational element;

V_j = the volume of the j^{th} water-column computational element, where j refers to the segment number;

Φ_{ij} = the source term for the state variable, B_i, in the j^{th} element; and

Γij = the sink term for the state variable, Bi, in the j^{th} element.

The benthic plant community state variables included in the model for the sediments are given in Table D-2.

D.2.1.3. *Vascular Macrophytes*

The hypothesis for the kinetics of vascular macrophytes is based on the terrestrial ecosystem energy model developed by O'Neill et al. (1972) for a closed-canopy, homogeneous forest ecosystem in the eastern deciduous biome. Bloomfield et al. (1973) adapted the concept to simulate aquatic macrophytes in Lake George, New York. The model for aquatic macrophytes in the Study Reach is similar to the Lake George model. Important features of this hypothesis are:

- The organic matter associated with vascular macrophytes can be idealized by three compartments for organic carbon: roots, leaves/shoots, and carbon storage.

- The accumulation of carbon in carbon storage is by photosynthesis. Carbon flows to roots and leaves/shoots from storage.

Other features of the model for vascular macrophytes are based on previous research and observations of macrophytes in the Middle Snake River (Falter and Carlson, 1994; Falter et al., 1995; Falter and Burris, 1996). These features are characterized by the following assumptions:

- Michaelis-Menten formulations are appropriate for light, nutrient, and habitat limitations (e.g., Barber, 1991; Porcella et al., 1983).

- Nutrient uptake rates are low at low river velocities because of poor rates of exchange, but increase with river velocity up to a certain optimum (Horner et al., 1983). As river velocity increase beyond a certain point, physical stresses begin to occur in the plants. These stresses lead to mortality of the plants and increase the rate of sloughing due to physical processes (Chambers et al., 1991a,b).

- Vascular macrophytes with extensive root systems, such as *Potamogeton,* take a large percentage of their nutrients from the sediments (Howard-Williams and Allanson, 1981). Macrophytes with limited root systems, such as *Ceratophyllum*, derive the majority of their nutrients from the water column.

D.2.1.4. *Epiphytes*

The hypothesis forming the basis for the model of epiphytes is a population model with Michaelis-Menten formulations for light, nutrient, and habitat limitations. This model was modified to include the assumption that in the Middle Snake River epiphytes such as *Cladophora* are generally associated with a macrophyte substrate on which they attach themselves and grow. Furthermore, the epiphytes intercept solar radiation in the top 10% of the water column, rather than over the entire water column. This assumption was based on observations made during the 1992-1994 studies of macrophytes (Falter and Carlson, 1994; Falter et al., 1995; Falter and Burns, 1996).

D.2.1.5. *Periphyton*

The hypothesis forming the basis for the model of periphyton is a population model with Michaelis-Menten formulations for light, nutrient, and habitat limitations. It is similar to those of Porcella et al. (1983) and Runke et al. (1981).

The mass balance equations for vascular macrophytes, epiphytes, and periphyton are given in Appendix B.

D.2.2. Variability and Uncertainty in Environmental Factors

Principal environmental factors for the Study Reach are hydrology and meteorology. The model accounts for variability and uncertainty of these environmental factors by assuming that the 67-year data record from 1928 to 1994 for river flow and air temperature for Twin Falls and Glenns Ferry is a representative sample for hydrology and meteorology. Use of the actual data record obviates the need for a complex time-series model, which would account for serial correlation between flows at various locations and times, serial correlation between air temperatures at various locations and times, and serial correlation between flows and meteorological conditions.

D.2.2.1. *Meteorology*

Meteorology data required for the analysis include air temperature, cloud cover, some measure of water vapor in the air such as dewpoint or relative humidity, and wind speed. Air temperature data for Glenns Ferry were used to characterize weather for segments of the Study Reach from Snake River Mile 547.0 to Snake River Mile 579.6. Air temperature data for Twin Falls were used to characterize water for the segments of the Study Reach from Snake River Mile 579.6 to Snake River Mile 640.0.

Estimates of daily values of dewpoint, cloud cover, and wind speed were made using a statistical analysis of data from the closest station with a long-term record for these variables. Daily values of these variables for each of the eight 3-hour periods in a day were assumed to be equal to the monthly averages for the eight 3-hour periods in a day for Burley, Idaho, as reported by the Pacific Northwest River Basin Commission (PNRBC) (1968).

D.2.2.2. *Hydrology*

The U.S. Geological Survey (USGS) maintains gaging stations at several locations in the Study Reach, including the beginning (USGS 13088000, Snake River at Milner) and ending (USGS 13154500, Snake River at King Hill) segments. In addition, it maintains gages at important surface inflows and springs. The records from these stations are very good and have been used to develop water budgets for the Snake River Plain in Idaho and Eastern Oregon (Kjelstrom, 1992). However, during the period in which flows have been recorded in the Snake River there have been numerous changes to the system. Major changes include the construction of dams for irrigation and power generation. Furthermore, the pattern of diversion for irrigation purposes has changed during this period. These changes have resulted in alterations of the hydrologic regime of the Snake River, in terms of both the magnitude and timing of flow (Richter, 1996).

Because of these changes in the hydrologic regime, the existing record of actual flows cannot be used directly to characterize risk associated with present management of the system hydrology. However, the Idaho Department of Water Resources (IDWR) has developed a model that estimates the historical monthly average flow and reach gain in the Snake River given present-day operating rules for the system. The results of applying the model to the period 1928-1994, reported as IDWR Study 150 (Robert Suter, IDWR, personal communication), are used to characterize hydrologic variability for the risk analysis in the Study Reach.

D.2.3. State Variables as Measurement Endpoints

In the framework for ecological risk, the focus in risk characterization is provided by assessment endpoints. The assessment endpoints represent the specific environmental goals to be addressed in the risk assessment. Measurement endpoints are quantitative estimates of the state of the ecosystem, which can be related in some way to the values expressed by the assessment endpoints. The assessment endpoints and their associated measurement endpoints are those shown in Table D-3. A logical source for measurement endpoints is the State of Idaho's water quality standards. These standards were developed in accordance with the Clean Water Act, in which the stated goal is "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." Those water quality standards that provide numeric criteria, or ones for which numeric criteria can be developed from narrative standards, are included in the analysis plan as measures of effect.

Although the water quality standards do provide measures of effect, they are typically measures that relate to a fairly broad range of aquatic species and environments. For example, the water quality standards define certain criteria for the protection of cold-water species without specifically describing which ones are the target organisms. These criteria may be protective of aquatic organisms within the group characterized as cold-water species, but there may be some organisms or certain life stages of organisms in this group that are not protected. Because of this it is desirable to develop site-specific lines of evidence if they are available.

For the Study Reach, there are site-specific measures of effect that can be integrated into the analysis plan to provide additional lines of evidence. Among these are indices the USFWS developed to characterize habitat suitability for certain cold-water fishes. The USFWS uses habitat suitability indices for assessing the impacts of flow modification on aquatic habitat resources of rivers and streams. An important objective of this method, called the Instream Flow Incremental Method (IFIM), is to make quantitative comparisons of habitat conditions at differing regimes of river flow.

Quantitative comparisons are accomplished in the IFIM by calculating habitat suitability for various regimes of river flow. Integral components of this calculation are the habitat suitability curves. These curves define suitability indices for different life stages of target aquatic species selected for a particular study. The suitability indices are functions of ecosystem state variables such as water depth, water velocity, water temperature, and substrate or cover type.

Because the indices measure suitability of habitat as a function of ecosystem state, they can also be used as measurement endpoints. However, the target species for which suitability is quantified must be relevant in terms of the assessment endpoints. For the Study Reach, this means they must be cold-water species that are native to the region. On the basis of the work of Anglin et al. (1992), three cold-water fishes found in the Study Reach, the mountain whitefish, rainbow trout, and white sturgeon, have been selected as target species to be used with the habitat suitability indices. Anglin et al. (1992) developed habitat suitability indices for these species for the Snake River from C.J. Strike Dam downstream to the upper end of Brownlee Pool. Because habitat suitability indices appropriate for large river systems were lacking for these species, Anglin et al. (1992) used criteria from smaller river systems. Extension of the criteria from smaller river systems to the Snake River was done subjectively based on the judgment of regional biologists. For the purposes of the ecological risk analysis, we have assumed the process adequately represents the habitat requirements for the target species in the Study Reach.

Performing ecological risk using simulation methods requires that measurement endpoints selected from the water quality standards and from the habitat suitability indices are also state variables simulated by the dynamic model (Tables D-1 and D-2). Those measurement endpoints for which state variables are simulated by the model and for which there are numeric water quality criteria and habitat suitability indices are given in Table D-4. The environmental values, or assessment endpoints, to which these quantitative measures are linked are also given in Table D-4.

The analysis of ecological risk using simulation methods addresses only a subset of the issues associated with ecological risk in the Study Reach. The hierarchy of interactions, from stressor source to stressor-to-stressor characteristics to endpoint for each of the assessment endpoints, as conceptualized in the Problem Formulation, is shown in Figure D-7. Those elements for which the simulation methods do not apply are shown as shaded and are primarily those affecting sediments and sediment transport.

D.2.4. System Boundaries, Length and Time Scales

D.2.4.1. *System Boundaries*

As defined previously, the Study Reach includes the segment of the Snake River from Milner Dam (RM 640.0) to King Hill (RM 545.3). The ecological risk analysis applies only to the main stem of the Snake River, but includes inputs of mass and energy from tributaries, irrigation returns, municipal waste discharge facilities, springs, and groundwater. The Study Reach was divided into two meteorological provinces, one from Milner Dam (RM 640.0) to Upper Salmon Falls Dam (RM 579.6), the other from Upper Salmon Falls Dam to King Hill (RM 545.3). The risk analysis also assumes that management rules for controllable input, both explicit and implicit, are based on existing conditions.

D.2.4.2. *Length Scales*

To obtain optimal spatial resolution with the mathematical model, the river was divided into several segments of the type shown in Figure D-5 (longitudinal orientation) for run-of-the-river reservoirs and freely flowing river segments and segments, of the type shown in Figure D-6 (vertical orientation) for those reservoirs exhibiting vertical stratification. Twin Falls and Shoshone Falls Reservoirs were treated as reservoirs with the potential for vertical stratification, on the basis of data collected by the Idaho Power Company (Myers and Pierce, 1996). For the run-of-the-river reservoirs and freely flowing river reaches, short segment lengths (0.1-0.3 miles) characterized the rapids, whereas the deeper, slower-moving sections between rapids were characterized by longer segment lengths (0.5-3.3 miles). Segments in the stratified reservoirs were chosen such that concentration gradients over vertical distances of 5.0 feet or more could be resolved.

For the risk analysis, it was useful to characterize the length scales for the river in terms of geomorphologic, hydrologic, and cultural features as shown in Table D-5. The rationale for these length scales is based on the work of Brockway and Robison (1992) and Covington and Weaver (1989, 1990a-c, 1991).

D.2.4.3. *Time Scales*

Time scales associated with the analysis are a function of the computational scheme used to solve the equations, the rates of response of ecological processes included in the model, and the time scales of the system inputs or driving forces. For purposes of obtaining an accurate solution in the mathematical sense, it is necessary to satisfy certain stability criteria required by the numerical methods used. These criteria are a function of the river flow, the rates of turbulent diffusion, and the volume of the river as described previously. The Study Reach is generally dominated by advection. For pure advection, the criterion for stability of the computational scheme is given by:

$$\Delta t_c < \frac{V}{Q}$$

where

Δt_c = the time increment for the numerical solution technique, seconds;
V = the net volume associated with the computational element, cubic feet; and
Q = flow into the computational element, cfs.

This time interval, Δt_c, is basically the time it takes a molecule of water to traverse a given segment. The numerical method that simulates the time history of mass and thermal energy computes the critical time increment necessary to satisfy this stability criterion. Therefore, the smallest time interval (highest frequency) for which state variables are simulated is a function of the river velocity and the segment length. For the Study Reach, this computationally driven time interval is smallest in the short, high-velocity segments (rapids) and can be as small as 100-200 seconds for some segments of the Snake River. In certain reservoir segments under low-flow conditions, this time interval can be on the order of days. For obtaining an accurate, stable solution to the set of difference equations, the integration time step must be equal to or less than the stability criterion given above.

The ecosystem processes included in the mathematical model (Equations A-1 to A-16 and B-1 to B-4) determine the time scales associated with ecosystem response. In this model, the ecosystem processes with the highest frequencies (shortest time scales) are those affected by diurnal variations in solar radiation. Ecosystem processes affected by diurnal variations in solar radiation are photosynthesis and the thermal energy budget. Therefore, the shortest time scales of ecological significance in the simulation results are those associated with the diurnal variations in solar radiation.

Other important time scales of simulated ecosystems processes are those associated with (1) rates of transfer of dissolved oxygen (deoxygenation and reaeration), (2) rates of transfer of the various forms of nitrogen and phosphorus (nitrification, mineralization, uptake by growth of aquatic plants), and (3) growth, respiration, and mortality rates of aquatic plants. The time scales for these processes are determined by the various rates of transfer. The time scales can vary from the order of hours to the order of months. The time scales of these rates are discussed in Section D.3. (Parameter Estimation).

The time scales of the system inputs or driving forces include, in addition to the diurnal variations in solar energy, annual variations in thermal energy budget and hydrology. Time scales associated with hydrology affect constituent loadings, as these loadings are also generally a function of hydrology. Available meteorology, hydrology, and constituent concentration data were used to characterize these time scales as described in Section D.3 (Parameter Estimation).

D.3. PARAMETER ESTIMATION

For analysis of ecological risk through simulation, it is necessary to estimate the parameters in the mathematical model as well as those that characterize the measures of ecological effects. The parameters in the mathematical model are those quantifying the rates at which mass and energy flow between state variables (Equations A-1 to A-16 and B-1 to B-4) and those quantifying the input or driving forces. Available data from the Study Reach and results of scientific studies in other river and lake systems provided the basis for estimating the parameters in the mathematical model. The State of Idaho's water quality standards and habitat suitability studies by the USFWS provided the basis for estimating the parameters used to characterize measures of ecological effects.

D.3.1. Ecosystem Driving Forces and Stressors

D.3.1.1. *Meteorology*

Air temperature, relative humidity or dewpoint, cloud cover, wind speed, and atmospheric pressure are required inputs to the model for estimating the heat budget and the amount of solar energy available for primary productivity. For the upstream segment (Milner Dam to Upper Salmon Falls Dam), meteorology was characterized by daily maximum and minimum air temperatures for Twin Falls using the data from Twin Falls Weather Service Office and monthly averaged wind speed, cloud cover and dewpoint for Gooding (PNRBC, 1968). For the downstream segment (Upper Salmon Falls Dam to King Hill), the meteorology was characterized by daily maximum and minimum air temperatures for Glens Ferry (National Weather Service's Local Climatological Summaries) and monthly averaged wind speed, cloud cover, and dewpoint for Gooding (PNRBC, 1968). Diurnal variations in air temperature were estimated from the maximum and minimum temperatures by assuming the temperature varied sinusoidally with a period of 24 hours. Incident solar radiation at the water surface was computed every 3 hours using air temperature, cloud cover, and solar altitude by the method described in WRE (1968).

D.3.1.2. *Hydrology*

The hydrology of this segment of the Snake River is complex. According to the USGS (1992), the springflow/groundwater contribution to the Snake River between Milner Dam and King Hill, Idaho, was approximately 4,000 cfs in 1980. In addition, there are major tributaries such as Rock Creek, Mud Creek, Deep Creek, and Salmon Falls Creek, as well as numerous irrigation return flows.

The water budget was derived from USGS daily flow data at Milner Dam (USGS 13088000), Kimberly (USGS 13090000), Buhl (USGS 13094000), King Hill (USGS 13154500); estimates of surface return flow given in Brockway and Robison (1992) and the State of Idaho's Division of Environmental Quality (DEQ) (1995); and estimates of the spring flows (Covington and Weaver; 1989, 1990a-c, 1991). The study area from Milner to King Hill was divided into four sections, the endpoints of each section being determined by the location of a USGS gaging station. For each section, estimates of the total surface return flows and spring flows were compared to the reach gain between gages as reported by the USGS. Differences between the total surface gains and the reach gain estimated from the gaging station were assigned to "groundwater" return flow. In addition, it was assumed that the groundwater flow between USGS gages, computed in this way, was distributed uniformly along the length of the river between the gages.

D.3.1.3. ***Hydraulic Characteristics***

The coefficients needed to define the hydraulic properties with relationships of the type described by Equations D-2 and D-3 were obtained from a regression analysis of surface water profiles simulated HEC-RAS (HEC, 1995). Cross-sectional characteristics of the river between Milner Dam and King Hill were obtained from a number of sources, including the U.S. Army Corps of Engineers, the Idaho Power Company, and local consulting firms (Gebhardt, 1992). Surface water profile analyses obtained using HEC-2 (HEC, 1991) by Brockway and Ralston (1992) were used for river segments between RM 610 and RM 607.0, whereas U.S. Army Corps of Engineers data were used for RM 607.0 to RM 588. Soundings of reservoir depth reported by Idaho Power Company were used to develop relationships for volume and area as a function of reservoir elevations for Twin Falls and Shoshone Falls reservoirs. Results of surface water profile analyses done for the Idaho Power Company were used to describe the hydraulic properties of the river segment between the Gridley Bridge and Bliss Dam.

An example of the calculation of the coefficients for velocity as a function of flow (Equation D-3) is given in Figure D-8. The example is taken from the Study Reach at RM 599.0. HEC-RAS was used to estimate average river velocities and water depths at flows of 1,500, 3,000, 5,000, 7,500, and 10,000 cfs. In the example shown in Figure D-8, a least-squares analysis of the log-transformed velocity and flow output from the HEC-RAS model provided estimates of the parameters, A_u and B_u. Using this method, coefficients for the relationship between velocity and flow and depth and flow were estimated for the other segments of the Study Reach. Hydraulic coefficients for Study Reach, estimated in this manner, are given in Table D-6.

D.3.2. Sources of Mass and Energy

There are a number of sources of energy and mass in the Study Reach. The sources included in the model are given in Table D-7. Water chemistry data collected by the University of Idaho Agricultural Research Station, Idaho DEQ, Clear Springs Food Inc., and the City of Twin Falls were used to estimate daily mass loadings from these point sources. The frequency at which the data were collected by each of these entities varied considerably from source to source. Much of the water quality and quantity data were collected from tributaries, irrigation returns, and fish hatcheries on a biweekly schedule during the period April through October and less frequently during the remaining months. The sampling frequency for the City of Twin Falls varied from daily for state variables such as flow, temperature, and pH to weekly for major nutrients. With the exception of the river flows measured at USGS gaging stations, very few measurements of water quality or quantity were made on a daily basis.

The numerical method that performs the simulations requires data at each computational time step. To accommodate this requirement, it was necessary to fill in missing data. In general, the method of filling in data was no more complex than linear interpolation between the times at which observations were made.

D.3.2.1. ***Hatcheries***

Long-term averages of water chemistry, from the U of I/ARS data set, collected during 1990-91, were used to characterize the mass loading from hatcheries. The average concentrations of flow and water chemistry for the hatcheries, as used in the analysis, are given in Table D-8. Daily estimates of mass loading, for input to the model, were estimated as being equal to the long-term values.

D.3.2.2. ***Major Tributaries***

East Perrine Coulee, Rock Creek, Cedar Draw Creek, Mud Creek, Deep Creek, Salmon Falls Creek, Billingsley Creek, and the Malad River were treated as major tributaries to the study section. Water chemistry used to characterize mass loading for these sources was obtained from various sources. Data source and period of record for each of these tributaries are shown in Table D-9. To obtain daily values for input to the model, missing data were filled in by linear interpolation when the data gap was less than 30 days. For data gaps greater than 30 days, the daily values were set equal to the monthly average, as computed from the rest of the record.

D.3.2.3. ***Minor Tributaries and Irrigation Return Flows***

Monthly averages of water chemistry, from the U of I/ARS data set (Brockway and Robison, 1992), were used for minor tributaries and irrigation return flows. Monthly averages of flow and water chemistry for the minor irrigation return flows are given in Table D-10. Daily values for input to the model were set equal to the average values for the appropriate month.

D.3.2.4. ***City of Twin Falls Sewage Treatment Plant***

Water quality and quantity data for Twin Falls sewage treatment plant were provided by the City of Twin Falls for the period 5/27/92 to 12/31/94. In addition, some measurements of quality and quantity were reported in the study conducted by Clear Springs Food, Inc. (MacMillan, 1992). For estimating model parameters, the entire available record was used, with linear interpolation between those days for which there were measurements filling in missing data.

For the ecological risk analysis, only that segment of data from the period 2/23/93-12/31/94 was used. The City of Twin Falls upgraded its treatment facility in late 1991-early 1992 (Nikki Arnold, EPA Region 10 Idaho Operations Office, personal communication) to increase the efficiency of solids removal. According to data provided by the City of Twin Falls (Figure D-9), an increase in ammonia removal efficiency did not occur, until early 1993 (see arrow in Figure D-9). The ammonia data from the beginning of February 1993 were therefore assumed to be representative of present operation of the sewage treatment facility. Means and standard deviations for various water quality and quantity of the sewage treatment plant effluent are given in Table D-11. These statistics were used to generate time series of effluent quality using the random walk model

$$X(t+1) = X(t) + N(0,\sigma^2) \qquad \text{(D-4)}$$

where

$X(t)$ = the value of some constituent, or its log transform, at the time, t; and

$N(0,\sigma^2)$ = a normal random deviation of mean, 0, and variance, σ^2.

Constituents with means and standard deviations of the log-transformed measurements in Table D-11 were modeled as the log transform then converted. Effluent temperature was modeled with the seasonal model

$$T(t) = \overline{T} + \Delta T \sin\left(\frac{2\pi(t - t_0)}{365}\right)$$

where

$T(t)$ = the daily effluent temperature, ºC;

$\overline{T}$ = the annual average effluent temperature, ºC;

ΔT = the annual temperature variation from the average, ºC;

t = the time, days; and

t_0 = the phase shift in the temperature, days.

D.3.2.5. *Spring Flows and Groundwater*

Water chemistry data collected by the USGS (USGS 13095500, Clark and Ott, 1996) and the University of Idaho Agricultural Research Station at Kimberly were used to estimate mass loadings from spring flows and groundwater. Springs included in the model are shown in Table D-12, as are the sources of water chemistry. The daily values of water chemistry input to the model were set equal to the long-term averages for the stations as shown in Table D-13. Daily values of water temperature were estimated from an equation of the form:

$$T(t) = \overline{T} + \Delta T \sin\left(\frac{2\pi(t - t_0)}{365}\right) \qquad \text{(D-5)}$$

where

$T(t)$ = the daily spring temperature, ºC;

$\overline{T}$ = the annual average spring temperature, ºC;

ΔT = the annual temperature variation from the average, ºC;

t = the time, days; and

t_0 = the phase shift in the temperature, days.

Parameters for Equation D-5 were estimated with nonlinear regression methods and are shown in Table D-14.

Water chemistry of the groundwater return flow, estimated as the difference between accumulated spring and surface return flow and actual flows measured at the USGS gaging stations, was assumed to be equal to the water chemistry of certain springs. The correspondence between water quality of the groundwater return flow and measured water quality of spring flow for each of the gaged segments is shown in Table D-15.

D.3.3. Dynamic Model of Mass and Energy Flow

In principle, the parameters for the equations (A-1 to A-16 and B-1 to B-4) of mass and energy flows can be inferred from properly designed field studies or observations. A properly designed field study is one in which all driving forces and environmental conditions are specified and there are enough measurements of state variables to formulate the problem properly. Because the parameter estimation process depends on the availability of data describing driving forces and the response of the river ecosystem to these forces, a comprehensive sampling program was designed jointly by the State of Idaho's DEQ and EPA Region 10. The sampling program included collection of water chemistry data in the Study Reach, tributaries, and point sources, as well as observations of biomass in the macrophyte, phytoplankton, and benthic invertebrate communities. The major data collection effort occurred during the period 1990-1994 and the results are reported in Brockway and Robison (1992, 1993), Falter and Carlson (1994), Falter et al. (1995), Falter and Burris (1996), Minshall et al. (1993), Minshall and Robinson (1994), and Royer et al. (1995). Data collected by Idaho Power Company (Ralph Myers, Shaun Parkinson, personal communication), Clear Springs Food (Macmillan, 1992), the City of Twin Falls (personal communication), and DEQ (Don Essig, personal communication) have also been used in the process.

When the problem is well defined, formal techniques (Menke, 1984) can be used to obtain estimates of parameters. Solving the parameter estimation problem generally involves minimizing some aspect of the difference between simulated and observed values. For example, the sum of the squared difference or, in some cases, the sum of the weighted squared difference between each simulated and observed value is commonly used. An estimate of the parameters is then obtained formally by solving the Equations A-1 to A-16 and B-1 to B-4 with the parameters as unknowns and the measured state variables as known. The solution, which minimizes the cost function, provides an estimate of the parameters. This process becomes extremely difficult as the number of parameters increases and as the nonlinearity of the governing equations increases.

The number of parameters is large in the Study Reach, the degree of nonlinearity is high, the number of measured state variables is limited, and the measurements are not without error. This makes it difficult to perform parameter estimation with formal methods. Because of the difficulty of applying formal techniques to solving the inverse problem, initial estimates of the parameters defining kinetics of mass and energy transfer for the water column were selected to be within the range of values used in similar models of surface water quality (Bowie et al., 1985). Selected parameters were then adjusted by trial and error to improve agreement between simulated and observed state variable estimates.

D.3.4. Rates of Mass and Energy Transfer

D.3.4.1. *Carbonaceous Biological Oxygen Demand (CBOD)/Dissolved Oxygen (DO)*

No data were available in the Study Reach to estimate the deoxygenation rate, K_1. Therefore, a value of $K_1 = 0.1$ days^{-1} (Equation A-1) was chosen for the entire river. CBOD from point and nonpoint discharges does not appear to be a major source of oxygen demand in the

Study Reach. Limited testing of the model showed that DO in the river was not sensitive to changes in this parameter within the ranges typical of this system (Bowie et al., 1985).

The reaeration rate, K2 (Equation A-2), was estimated in a number of different ways, depending on the hydraulic properties of each reach. For the two reservoirs, Twin Falls and Shoshone Falls, the formulation for reaeration coefficients in lakes developed by Smith (1978) was used. In run-of-the-river segments where the water was moving relatively slowly, the O'Connor-Dobbins (1958) formulation was applied. In run-of-the-river segments where the water velocities were higher, such as Auger Falls, Boulder Rapids, Empire Rapids, and Kanaka Rapids, the Churchill et al. (1962) formulation was applied.

D.3.4.2. *Phytoplankton Biomass*

Rates of phytoplankton growth, respiration, nutrient, light and temperature limitations, and stoichiometry were initially based on values typical of those used in other phytoplankton model studies (Bowie et al., 1985). Sensitivity analysis showed the dynamics of phytoplankton in the Study Reach to be more responsive to hydrology and to initial conditions from upstream sources.

D.3.4.3. *Nitrogen*

Rates of flow of nitrogen between the species, organic N, NH_4-N, and NO_2+NO_3-N were initially estimated as being in the range found in other studies (Bowie et al., 1985). In the Auger Falls segment below the City of Twin Falls STP, the nitrification rate, K_{55}, was increased from the initial estimate of 0.25 days^{-1} to 2.25 days^{-1} to achieve better agreement between simulated and observed values.

D.3.4.4. *Phosphorus*

The rate of mineralization for organic phosphorus was estimated from the literature (Bowie et al., 1985) as 0.005 days^{-1} throughout the entire Study Reach. This corresponds to a time constant of approximately 200 days. The residence time of a water parcel in the Study Reach is much less than this. The simulation results are, therefore, not sensitive to changes in this parameter.

Initial simulations resulted in a consistent overestimate of total phosphorus at all locations in the Study Reach. Unbiased simulations of phosphorus were obtained by incorporating a loss rate for organic and inorganic phosphorus into the mass balance equations (Equations A-13 and A-15). The rates were estimated to be 0.5 days^{-1} and 0.05 days^{-1} for organic and inorganic phosphorus, respectively. Sediment trap data reported by Falter and Burris (1996) provided the basis for these estimates.

D.3.4.5. *Vascular Macrophytes and Epiphytes*

The mathematical model for vascular macrophytes and epiphytes (Equations B-1 to B-4) is a hypothesis based on previous work in lakes (Bloomfield et al., 1973) and the eastern deciduous forest biome (O'Neill et al., 1972). The model is highly nonlinear and contains many parameters. Extensive field studies of aquatic macrophytes were conducted in the Study Reach (Falter and Carlson, 1994; Falter et al., 1995; Falter and Burris, 1996). However, given the complexity of the model, it was necessary to estimate many of the parameters from the literature and then to adjust the most sensitive parameters until there was reasonable agreement between simulated and observed state estimates.

The extensive macrophyte literature provided a basis for the initial parameter estimation process. MacMillan (1992) has a comprehensive discussion of macrophyte and epiphyte kinetics by Barber (1991), and Krousel (1991) reported values for growth rates, respiration rates, Michaelis-Menten half-saturation constants for nitrogen and phosphorus, and mortality rates in three species of vascular macrophytes and epiphytes. van Wijk (1989) measured Michaelis-Menten half-saturation levels for phosphorus and nitrogen in *Potamogeton pectinatus*. Chambers et al. (1991a,b) measured the response of macrophytes to river velocity and to nutrient concentrations in the sediments and the water column. Horner et al. (1983) developed a mechanism-based model of periphyton growth based on river velocity and nutrient enrichment.

Initial estimates of parameters describing macrophyte and epiphyte kinetics were derived from these studies. Many of the initial estimates came from the work of Barber (1991), given the similarities between Barber's model and the one used in this analysis.

In addition to the parameters characterizing mass and energy transfer, a benthic habitat factor was introduced. The benthic habitat factor was an estimate of the fraction of the bottom area available for macrophyte growth in each river segment. Downstream of Auger Falls (RM 606.6) this factor was estimated from the macrophyte studies conducted by Hill (1992). Above Auger Falls the habitat factor for macrophytes was assumed to be zero, primarily because of a lack of data. The habitat factors used for each segment are given in Table D-16.

Initial estimates for the parameters characterizing growth rates, rates of senescence, and nutrient uptake were varied by trial and error using mass and energy loading as described, 1990-1994 water chemistry and hydrology data described above, and 1992-1994 macrophyte data reported by Falter and Carlson (1994).

Parameters in the surface water and benthic systems equations were varied until the outcomes appeared as shown in Figures D-10 to D-16. Parameters for the model of surface water quality are given in Table D-17 and for the model of benthic plants are given in Tables D-18-D-20. Because the parameter set for the system of equations is so large and the database limited, the resulting parameter set (Tables D-17 to D-20) may not be unique. That is, there may be other parameter sets that lead to results similar to those obtained in this initial test.

D.3.5. Results

The mathematical model described above was used to simulate state variables (Tables 1 and 2) in the Snake River between RM 640.0 and RM 545.5 for the period January 1, 1990, to December 31, 1994. The simulated surface water quality results are compared with water quality data collected by the University of Idaho's Agricultural Research Station, Idaho State University, and Clear Springs Foods, Inc., at various locations in the Snake River (Table D-21) in Figures D-10 to D-16. Means and standard deviations of the differences between simulated and observed values at these locations are shown in Table D-22.

Comparison of simulated and observed results for those days on which measurements of macrophytes and epiphytes were made by the University of Idaho (Falter and Carlson, 1994; Falter et al., 1995; Falter and Burris, 1996) are shown in Figures D-17-D-19. Simulated results are for laterally averaged values of the two macrophyte types (*Potamogeton* and *Ceratophyllum*) and the epiphyte (*Cladophora*).

D.3.6. Discussion

D.3.6.1. *Dissolved Oxygen*

Differences in simulated and observed DO are biased in the direction of underprediction (simulated values less than observed, on average) at the upstream stations (S04, S13, S21, and S31) and biased in the direction of overprediction at the downstream stations (S40, S49, and S72). The differences are generally independent of the magnitude, although the model consistently underpredicts the maxima at Stations S13, S21, S31, S40, and S49. These maxima appear to be related to periods of high primary productivity. Some of this difference may therefore be due to the simulated results being estimated as daily-averaged values whereas the observations are point measurements at a particular instant in time.

D.3.6.2. *Water Temperature*

Differences in simulated and observed water temperatures are biased in the direction of underprediction at all stations except S40 (Table D-22). The average difference between simulated and observed water temperature at S49 is not significantly different from 0.0. With the exception of the two upstream stations (S04 and S13), the difference between simulated and observed is negative at water temperature below about 12 ºC and positive above (Figures D-20b to D-26b). This bias is most likely due to incomplete knowledge of the inflow temperature from nonpoint sources, particularly springs.

An important characteristic of the water temperature in the Study Reach, as reflected in both the simulated and observed values, is the change in temperature range from upstream to downstream. At the location farthest upstream (RM 612), water temperature varies from near 0 ºC to approximately 22 ºC. At the downstream locations the water temperature ranges from approximately 7 ºC to 20 ºC. The difference in water temperature range is due to the moderating effects of the spring flow on river temperatures.

D.3.6.3. ***Ammonia Nitrogen***

Bias in the differences between simulated and observed values of ammonia nitrogen shows no consistent pattern (Table D-22). Bias is highest at station S13, which is just below the Auger Falls reach of the Snake River. However, the degree of correlation between model and observed is low at stations S40, S49, and S72. This implies that the kinetics of the model may not properly represent important processes found in the Study Reach (Figures D-24c to D-26c), particularly in those areas for which primary productivity is high.

D.3.6.4. ***Nitrate Nitrogen***

Differences between simulated and observed values of nitrate nitrogen are biased in the direction of underprediction at the upstream stations (S04, S13, and S21) and in the direction of overprediction at the downstream stations (S31, S40, S49, and S72) (Table D-22). The variance of the difference decreases in a downstream direction. The range of nitrate nitrogen also decreases in a downstream direction. At S04 the range is from less than 0.5 mg/L to more than 2.5 mg/L, whereas at S72 it is from about 1 mg/L to 2 mg/L. As in the case of temperature, this change can be attributed to the moderating effect of the springs on both flow and concentration.

D.3.6.5. ***Total Phosphorus***

Initial simulations of total phosphorus gave results that were biased in the direction of overprediction at the upstream locations and in the vicinity of the maximum macrophyte density (Stations S04, S13, S21, and S31). That is, the simulated levels of total phosphorus were generally higher than the observed levels of total phosphorus. The model appears to be either underpredicting phosphorus uptake by aquatic macrophytes or failing to account for deposition of phosphorus associated with sediments or particulate matter. Using sedimentation rates for phosphorus reported by Falter and Burris (1996) gave results that were nearly unbiased, as shown in Figures D-20e to D-23e and Table D-21. Model simulations at the downstream stations (Stations S40, S49, and S72) are biased only slightly in the direction of overprediction (Figures D-24e to D-26e). The standard deviation of the model error is highest above Snake River Mile 581 (Upper Salmon Falls Dam), where primary productivity is also high.

D.3.6.6. ***Vascular Macrophytes and Epiphytes***

Simulated and observed macrophyte biomass values are plotted for calendar years 1990-94 at two locations in the MSR, RM 600 and 589 (Crystal Springs and Box Canyon, respectively) (Figures D-17 to D-19). Separate plots of macrophyte density (g C oven-dry weight/m^2) are given for rooted macrophytes, nonrooted macrophytes, and epiphytes.

D.3.6.7. *Crystal Springs (RM 600)*

In 1992, predicted rooted and nonrooted macrophytes were three to four times greater than observed values. Predicted epiphytes closely matched observed, but predicted levels of epiphyte demise lagged observed by about 1 month. The answer could well lie in the very early development of epiphytes in 1992. By early June, epiphytes were at annual peak levels, 4 to 6 weeks ahead of 1993 and 1994 peaks (Falter et al. 1994). The high temperatures and very low flows of early 1992 are a likely reason for the early epiphyte development in those years, as water temperatures were 22 ºC-24 ºC by mid-June. In 1993 and 1994, however, mid-June water temperatures were 4 ºC-5 ºC cooler than in 1992. Higher and later spring flows in 1993-94 were the cause of cooler temperatures those years. It is likely that the warm, low flows of 1992 favored early epiphyte development, which suppressed rooted and nonrooted macrophytes by blanketing, light suppression, and nutrient competition through the first half of the summer. These biological interactions were not captured by the model. The likely cause for epiphyte demise earlier than predicted is not apparent.

In 1993, predicted rooted macrophytes closely modeled observed, both in magnitude and timing. Predicted nonrooted macrophytes were slightly lower and about a month later than observed. In 1993, water flows were much different than in 1992. Flows had a May flood pulse of 14,000 cfs and a midsummer pulse of about 2,100 cfs, compared with 1992 midsummer flows of 350 cfs. These higher flows were 4 ºC-5 ºC cooler than in 1992 and could have favored both rooted and nonrooted macrophytes. Observed values of these forms were several-fold greater than 1992 values. Optimal growth temperatures in the model were set at 25 ºC for both rooted and nonrooted forms. It is possible that nonrooted optimal growth temperature should be set lower to project greater growth at cooler temperatures. (It should be noted that modeled nonrooted growth in Box Canyon reach, where water temperatures were much cooler because of springs influence, was also much less than observed growth.) Early growth of observed epiphytes was suppressed in 1993 and did not peak more than a few months after 1992 observed epiphyte peaks. The model correctly predicted epiphyte levels, but predicted levels declined in the fall about 2 months before the observed epiphyte decline (Figure A-18).

In 1994, there was no spring flood pulse, but flows through the summer and fall held at high summer flows ~60 m^3/s (compared with 10 m^3/s in 1992 and 10 m^3/s with an early summer pulse in 1993). Early summer temperatures were about 6 ºC-7 ºC cooler than in 1992. The onset and peaks of rooted macrophytes were correctly predicted; the prediction missed, however, the observed midsummer decline and low rooted levels through the fall. Nonrooted predictions more closely matched observed, both in levels and timing (Figure D-17). Predicted epiphyte peaks and timing matched observed but declined about a month and a half sooner in the fall than observed.

D.3.6.8. *Box Canyon (RM 589)*

Predicted and observed macrophyte comparisons are available only for 1992. As in Crystal Springs, rooted growth was overpredicted, but only by ~50%. Nonrooted growth was underpredicted by about the same amount. In both cases, predicted peaks lagged observed by about 2 months. This lag was also observed in Crystal Springs in 1992. Epiphyte biomass peaks and timing were reasonably predicted at Box Canyon in 1992.

D.3.6.9. *Measurement Endpoints*

The relationships among state variables, measurement endpoints, and assessment endpoints are shown in Table 4. The parameters characterizing the measurement endpoints are derived from the State of Idaho water quality standards and from habitat suitability curves developed for certain cold-water fishes native to the Snake River. As discussed in Chapter 3 (Conceptual Model), the target organisms chosen for the Study Reach are the rainbow trout, mountain whitefish, and white sturgeon. Therefore, where the measurement endpoints are in terms of specific organisms, the target organism was selected from this group of three.

D.3.7. State of Idaho Water Quality Standards

For cold-water biota, the water quality standards of the State of Idaho require the waters of the Study Reach to have the following characteristics.

D.3.7.1. *Dissolved Oxygen*

Dissolved oxygen concentrations must exceed 6 mg/L at all times, with the following exceptions:

- The bottom 20% of water depth in natural lakes and reservoirs where depths are 35 meters or less.
- The bottom 7 meters of water depth in natural lakes and reservoirs where depths are greater than 35 meters.
- Those waters of the hypolimnion that are in stratified lakes and reservoirs.

For salmonid spawning, the State of Idaho water quality standards require the one-day minimum dissolved oxygen be not less than the greater of 6.0 mg/L or 90% of saturation during the spawning period and incubation period for the particular species inhabiting the waters. Time periods for spawning and incubation of species native to the waters of Idaho are given in Table D-23.

D.3.7.2. *Water Temperature*

For cold-water biota, the water quality standards of the State of Idaho require the waters of the Study Reach to have water temperatures of 22 °C or less with a maximum daily average of no greater than 19 °C. For salmonid spawning, water temperatures must be equal to or less than 13 °C, with a maximum daily average no greater than 9 °C during the spawning period and incubation for the particular species inhabiting the waters of the Study Reach.

D.3.7.3. *Ammonia*

Criteria for toxicity of un-ionized ammonia to coldwater species in the Study Reach are shown in Tables III and IV of the State of Idaho's water quality standards. Criteria are given as a function of pH and water temperature for both chronic and acute toxicity.

D.3.7.4. *Excess Nutrients*

The State of Idaho's water quality criteria for nutrients are in a narrative form:

> "Surface waters of the state shall be free from excess nutrients that can cause visible slime growths or other nuisance aquatic growths impairing designated beneficial uses."

For purposes of establishing a Total Maximum Daily Load (TMDL) for total phosphorus in the Study Reach, the State of Idaho's DEQ (1995) interpreted these narrative water quality criteria in the Study Reach in terms of a numeric criterion. The TMDL requires that total phosphorus be 0.075 mg/L when the river flow is equal to the 1-in-10-year 7-day average low flow (${}_7Q_{10}$), as measured at the Gridley Bridge (Snake RM 583.0). This criterion is less than that suggested in U.S. EPA (1976) for flowing waters (0.10 mg/L), but greater than that suggested by U.S. EPA (1976) for flowing waters that enter lakes or reservoirs (0.05 mg/L). The criterion for total phosphorus developed by Idaho DEQ has not been specifically related to the levels of vascular macrophyte growth in the river that would exceed the State of Idaho's narrative water quality standard for nutrients. The State of Idaho's DEQ (1995) identified aquatic macrophyte growth as being present at high or nuisance levels. In an effort to define a measurement endpoint for nuisance levels of aquatic macrophyte, a literature survey was conducted. Only those papers that made reference to water quality impacts and that had quantitative data for macrophyte biomass were used to develop the measurement endpoint. Types of water quality impacts included general water quality degradation, alteration of the aquatic environment, and eutrophic conditions. The results of this survey (Table D-24) suggest that 200 g/m^2, average maximum biomass as AFDM, is a reasonable lower bound for nuisance levels of aquatic macrophytes.

D.3.8. Habitat Suitability Curves

The U.S. Fish and Wildlife Service, in support of the IFIM, has developed habitat curves for many aquatic organisms. In a cooperative study conducted with the Idaho Power Company, Anglin et al. (1992) described habitat suitability curves for the three cold-water fishes identified as target organisms for the Study Reach. The suitability indices used in this study for spawning, incubation, fry/larvae, juvenile, and adult stages of the mountain whitefish, rainbow trout, and white sturgeon are shown in Figures D-27 to D-38. The periods of the year to which these suitability indices apply are shown in Table D-25.

In applications of IFIM, the habitat suitability curves are used to develop flow-weighted measures of habitat suitability. In this ecological risk assessment, the simulation results from the ecological model were compared to the IFIM habitat suitability curves. The frequency with which the simulated results were less than a reasonable value of the habitat suitability curve was used to assess whether or not the system would support a particular life stage of the target organism. The reasonable level for habitat suitability was defined as 0.6. That is, values of the habitat index greater than 0.6 in Figures D-27 to D-38 were assumed to represent conditions supporting the particular life stage of the target organism, and values of the index less than 0.6 were assumed to represent conditions that would not support that life stage. The criterion of 0.6 was chosen simply because it is slightly greater than 0.5. Although this choice was somewhat arbitrary, the estimates of ecological risks are not particularly sensitive to the criterion, given the shapes of the habitat suitability curves (Figures D-27 to D-38). Most of the uncertainty in estimates of ecological risks using these habitat suitability curves is in the shapes of the curves.

D.4. EXPOSURE ANALYSIS

The objective of this analysis is to characterize elements of exposure by simulating the space/time distribution of state variables that are also measures of effects to the ecosystem. The simulations are performed with the dynamic model of mass and energy balance. Environmental driving forces are input to the model with measures of variability and uncertainty developed from the historical record. The simulated state variables will have uncertainty and variability that is representative of the Study Reach, assuming the models for driving forces and for mass and energy balance are representative of the processes in the Study Reach.

Limitations in our understanding of ecosystem processes in the Study Reach are such that the model does not simulate all the state variables that characterize the primary stressors described in the Problem Formulation. In particular, the model does not include those state variables necessary to characterize sediment loading and habitat alteration associated with changes in the substrate. The state variables simulated by the model that are primary stressors and can be used to measure effects of exposure are listed in Table D-4. The quantitative analysis of ecological risk, within the framework of the simulation methods described in this study, addresses only those stressors shown in Figure D-3.

D.4.1. Stressor Characterization

The stresses to the Study Reach ecosystem are a result of hydrologic modification and the input of nutrients from point and nonpoint sources. The stressors for this analysis have been characterized in terms of the existing conditions. Existing conditions for the analysis are described in Table D-26.

D.4.2. Temperature-Dissolved Oxygen

Ecological integrity of an aquatic ecosystem is dependent on the characteristics of the water temperature and dissolved oxygen regimes. The State of Idaho water quality standards for the Study Reach protect beneficial uses associated with cold-water species and spawning of cold-water fishes. For characterizing stress associated with temperature and dissolved oxygen, the simulated 67-year record of state variables is compared to the water quality criteria in each of the representative segments. The comparison is made for the general category of cold-water species and for spawning of two species, the mountain whitefish and the rainbow trout. The temperature-dissolved oxygen state-space diagram, which plots the temperature as a function of dissolved oxygen, provides a compact format for making the comparison.

Stress occurs when the temperature-dissolved oxygen envelope experienced by a target organism is larger than the envelope associated with its physiological requirements. Superimposing the envelope for water temperature and dissolved oxygen given in the water quality standards for each of these groups on the simulated temperature-DO state-space diagram is a way of assessing stress associated with the temperature-DO regime in a particular segment (Figures D-39 to D-51). The frequencies with which the simulated values fail to fall within the envelope for temperature and DO defined by the State of Idaho's water quality standards are given in Table D-27.

For cold-water biota the frequency with which the simulated daily-average DO falls outside the envelope defined by the water quality standards is less than 0.01 in all of the segments. Frequencies for spawning mountain whitefish are less than 0.06 throughout the Study Reach, except in the segment from Kanaka Rapids to Gridley Bridge, where the frequency is 0.11. The frequencies for spawning rainbow trout range from 0.26 to 0.57 between Milner Dam and the Gridley Bridge, and 0.19 to 0.45 between the Gridley Bridge and King Hill. The higher frequencies associated with spawning rainbow trout are due to the fact that applicable water quality standards include some summer months, whereas the water quality standards for spawning mountain whitefish apply to fall and winter months when saturation levels of DO are higher.

The frequency with which the daily-average simulated water temperature falls outside the envelope for cold-water biota ranges between 0.13 and 0.20 for the segments between Milner Dam and Kanaka Rapids. The frequency decreases to less than 0.05 from Kanaka Rapids to the Bliss Bridge because of the large volume of cooler water supplied by the spring flow. Between the Bliss Bridge and King Hill the frequency increases slightly as the spring flow decreases and the transfer of thermal energy across the air-water interface becomes more important.

The pattern for the frequency with which the simulated maximum water temperature falls outside the envelope for cold-water biota is similar to that of the simulated daily-average water temperature. The decrease in frequency occurs in the Crystal Springs to Boulder Rapids segment slightly upstream from Boulder Rapids to Kanaka Rapids, the segment in which the frequency of daily-average water temperatures decreases. In addition, the magnitude of the frequencies for which the maximum temperatures fall outside the envelope for cold-water biota is less than for the daily-average temperatures. The frequencies for maximum simulated water temperatures varied between 0.01 and 0.13 from Milner Dam to Boulder Rapids and were less than 0.01 between Boulder Rapids and King Hill.

The frequencies with which simulated daily-average water temperatures fall outside the envelope for spawning rainbow trout is greater than 0.58 throughout the Study Reach and greater than 0.48 for simulated maximum daily water temperatures. The high frequency with which DO falls outside the envelope is due to the fact the water quality standards for spawning rainbow trout include some summer months.

For water quality standards applicable to the spawning of mountain whitefish, the frequency with which water temperature falls outside of the envelope is reversed compared to cold-water biota. Frequencies with which daily-average and maximum water temperatures fall outside the envelope are lowest in the upstream segments between Milner Dam and Kanaka Rapids and highest in the segments between Kanaka Rapids. Frequencies for daily-average simulated water temperatures vary between 0.05 and 0.10 in the upstream segments and between 0.10 and 0.22 in the downstream segments. The reversal in the pattern and the relatively low frequency of exceedances is due to the fact that water quality standards for spawning mountain whitefish apply to fall and winter periods when water temperatures are low.

D.4.3. Total Phosphorus/Macrophytes

Phosphorus in sufficiently high concentrations is a stressor inasmuch as it promotes the growth of undesirable aquatic plants and large blooms of phytoplankton. Vascular macrophytes in segments of the Study Reach have caused the loss of beneficial water uses. An environmental goal of watershed management, and an assessment endpoint in the Study Reach is to reduce the density of macrophytes and levels of phytoplankton. A major element in this goal is the reduction of phosphorus from both point and nonpoint source discharges.

The State of Idaho's DEQ has developed a TMDL for total phosphorus in the Study Reach as part of the management plan to reduce macrophyte density. The TMDL limits total phosphorus in the Study Reach at the Gridley Bridge (RM 583.0) to 2,400 lb/day. This corresponds to a concentration of 0.075 mg/L total phosphorus for a flow of approximately 5,900 cfs in the Snake River.

The simulation results from the dynamic model were used to obtain an empirical cumulative distribution function (CDF) for total phosphorus at a location representing the segments given in Table D-5. The CDFs for total phosphorus are shown in Figures D-52 to D-64. Uncertainty bands for the CDFs in Figures D-52 to D-64 reflect the standard deviations for the differences between simulated and observed total phosphorus as shown in Table D-26. The empirical CDF gives the probability that the simulated total phosphorus is equal to or less than some specified value. These empirical CDFs represent stressor characteristics for existing levels of management and do not incorporate the limits specified in the TMDL.

In the segments of the Study Reach upstream from major point source and nonpoint source inputs, the estimated probability that total phosphorus will be equal to or less than 0.075 mg/L is between 0.23 and 0.25. Total phosphorus loads from the City of Twin Falls STP, fish hatcheries, and irrigation return flows reduce the probability that total phosphorus will be equal to or less than 0.075 to between 0.01 and 0.04 in the segments between Shoshone Falls and Gridley Bridge. The large volume of spring inflow with low levels of total phosphorus to the Study Reach increases the estimated probability to 0.07-0.18 between Gridley Bridge and King Hill.

The cumulative distribution function for total macrophyte and epiphyte biomass was estimated for the Crystal Springs to Boulder Rapids segment. This segment was chosen because it has had among the highest levels of macrophyte growth measured during field studies by the University of Idaho (Falter et al., 1995; Falter and Burris, 1996). The probability that the simulated values of macrophyte biomass, measured as the sum of rooted macrophytes, nonrooted macrophytes, and epiphytes, would be less than 200 gm C AFDM/m^2 was estimated to be less than 0.01 for the 67-year period of record (Figure D-65).

D.4.4. Un-Ionized Ammonia

Simulated levels of un-ionized ammonia were compared to the chronic and acute criteria in Tables III and IV of the State of Idaho's water quality standards. The frequency with which the simulated levels exceeded the criteria was computed as the ratio of the number of values that exceeded a criterion divided by the number of simulated values.

The estimated frequency with which the simulated values of un-ionized ammonia are below the chronic and acute criteria was estimated to be less than 0.05 and 0.01, respectively, throughout the Study Reach.

D.4.5. Habitat Suitability Curves

Measures of stress to the target cold-water species, mountain whitefish, rainbow trout, and white sturgeon, due to water temperature and hydrologic effects were obtained from habitat suitability indices developed by the USFWS (Anglin et al., 1992). The measures of habitat suitability associated with water temperature, water depth, and velocity were obtained for each life stage of the target organism and determined in the following way. Initially, a value of 0.6 was chosen as a limiting value that would apply to both upper and lower limits of habitat suitability for all habitat factors (water depth, water velocity, and water temperature), all life stages, and all species. For each of the suitability curves (Figures D-27 to D-38), when the simulated value of the habitat factor resulted in an index value less than 0.6, the condition of the water in the simulated segment was characterized as impaired for that factor, life stage, and target organism. For simulated values of the habitat factor resulting in an index value greater than 0.6, the condition of the habitat was characterized as unimpaired, or impaired if the value was less than 0.6.

The index of impairment for each life stage was computed as the ratio of the number of simulated days in which impairment of habitat occurred to the total number of days for which the life stage was vulnerable (Table D-23). Four categories of impairment were defined on the basis of the index of impairment. The results are shown in Figures D-66 to D-68 for the various life stages of the target organisms: rainbow trout, mountain whitefish, and white sturgeon.

The index of impairment was generally high throughout the Study Reach for all life stages of rainbow trout. Estimated levels of impairment for adults were estimated to be moderate to low in some portions of the segments from Rock Creek to Crystal Springs, from Boulder Rapids to Kanaka Rapids, and in the upper portion of Upper Salmon Falls reservoir. Estimated levels of impairment for spawning were low to moderate in some portions of the segment from Boulder Rapids to Kanaka Rapids, as were estimated levels of impairment for rainbow trout fry.

Estimated values of the index of impairment for spawning, fry, and juvenile mountain whitefish were high throughout the Study Reach. For adult mountain whitefish, the values of the index of impairment were estimated to be moderate to high throughout the Study Reach. The most favorable conditions for adult mountain whitefish (moderate impairment) were found in Upper Salmon Falls and Lower Salmon Falls reservoirs. These somewhat favorable conditions were primarily a result of high-quality cooler water entering the Study Reach from springs.

Above Lower Salmon Falls Dam, estimated values of the index of impairment for white sturgeon were generally high for all life stages. Exceptions were found in the pool below Auger Falls and in Upper Salmon Falls reservoir, where the index of impairment for adult white sturgeon was estimated to be low and the index of impairment for juveniles was estimated to be moderate. Portions of the Wiley reach (Lower Salmon Falls Dam to the Bliss Bridge) had low to moderate values of the index of impairment for spawning and larval stages. The most favorable conditions for all life stages of white sturgeon were estimated to occur in the Study Reach below Bliss Dam.

D.4.6. Uncertainty and Variability

Ecological risk is the likelihood that adverse ecological effects may occur as a result of exposure to one or more stressors (U.S. EPA, 1992) The primary goal of ecological risk assessment is to identify and reduce variance in characterizing the response of ecosystems to stressors. This variance arises from variability and uncertainty. The taxonomy of variability and uncertainty in risk assessment has been discussed by a number of authors. Suter (1990), for example, identified the following three sources of uncertainty and variability: the inherent randomness of the world (stochasticity), imperfect or incomplete knowledge of things that could be known (ignorance), and mistakes in execution of assessment activities (error). For this risk assessment, using simulation methods, sources of uncertainty and variability include

- variability and uncertainty in ecosystem driving forces and stresses,
- variability and uncertainty in sources of mass and energy,
- model error,
- parameter estimation error,
- state variable measurement error, and
- variability and uncertainty in measurement endpoints.

D.4.6.1. *Ecosystem Driving Forces*

The models for characterizing the variability of the ecosystem driving forces, hydrology and meteorology are described in Section 4.1. The models for variability in both hydrology and meteorology are based on the actual 67-year record of the meteorology and the adjusted 67-year record of the hydrology. Even though using the actual record captures some of the variability in a simple and straightforward manner, the record represents only a sample of the actual population. As such, it may not include events that have a low probability of occurrence. Such events would likely be associated with long-term changes in regional and global weather patterns.

In addition to the variability in the ecosystem driving forces, there is uncertainty in the models for these forces. In the case of hydrology, there is uncertainty in the model IDWR uses to account for water management in the Snake River basin. There is also uncertainty in the way in which the monthly flows estimated by the IDWR hydrologic model were disaggregated to the daily values. For the meteorology, there is uncertainty in the spatial variability of weather despite the fact that an effort was made to account for some of this by using two weather stations. The meteorologic variables most likely to be affected by the uncertainty in spatial variability are wind speed and air temperature. Although the energy budget of the Snake River in the Study Reach is dominated by advection, the ecological risk analysis would benefit from a reduction of uncertainty in wind speed and air temperature.

D.4.6.2. *Sources of Mass and Energy*

With the limited data available, it was difficult to account for or separate uncertainty and variability in the sources of mass energy. The longest record available for developing models for the sources of mass and energy was 5 years (1990-1994). In addition, sampling periods for these sources were generally biweekly or greater. The data gaps were much larger for the springs, which play an extremely important role in the Study Reach. The relatively low variability of both the quantity and quality of the spring flows mitigates the impact of these data gaps to a degree. For sources with important high-frequency components such as irrigation return flows, the data gaps are likely to be significant. In general, the limited data imply a high degree of uncertainty for both high-frequency and low-frequency components of the sources of mass and energy.

D.4.6.3. *Model Error*

Errors in the simulation model for ecological risk analysis (Appendices A and B) contribute to uncertainty in the estimate of ecological risk in a number of ways. Errors in structure are generally the major sources of model uncertainty in ecological models, and result from omitting important state variables or flow paths between state variables.

Spatial and temporal aggregation of state variables in the simulation model also contribute to model uncertainty. The equations of mass and energy balance for this model assume the state variables describing the water body vary only vertically or longitudinally, depending on the nature of the water body. In addition, the simulated state variables represent volume averages for river segments 0.3 to 3.3 miles in length and reservoir segments 5.0 feet thick. State variables are also averaged over a 3-hour period. There will therefore be uncertainty regarding the system state at spatial scales smaller than the segment size and at time scales less than 3 hours.

Aggregation of state variables in space and time can result in errors generated by the method used to solve the finite difference equations (Appendices A and B). These errors are largest where there are strong spatial gradients in state variables and where the time increment for the numerical solution technique, Δt_c, is not equal to time required for a molecule of water to traverse a given segment.

All of these errors in model structure can give rise to uncertainty in the ecological risk analysis. As is the case for most ecological models, the structure for the ecological model used in this analysis of risk is a hypothesis derived from previous ecological model construction and field studies in rivers, lakes, and reservoirs, including the field studies done in the Study Reach. There are at this point no widely accepted protocols for testing hypotheses regarding state-space models of the type used in this analysis. Oreskes et al. (1994) suggest a qualitative comparison of model simulations with observed values of the state variables as a way of establishing the credibility of earth science models.

Simulated results for DO, water temperature, total ammonia nitrogen, nitrite-nitrate nitrogen, and total phosphorus are plotted against observed values of these same state variables in Figures D-20 to D-26. These plots show that the best correlation between simulated and observed state variables occurs for water temperature and for nitrite-nitrate nitrogen. Correlation between simulated and observed values for DO is generally good, with the exception of the first part of 1992, when supersaturated levels of DO were observed (Minshall et al., 1993) but were not simulated by the model. This implies the model structure may not include all the processes required to accurately simulate primary productivity in the Study Reach.

Correlation between simulated and observed values of total ammonia nitrogen and total phosphorus was generally low. The total ammonia nitrogen results showed significant positive bias (the model predicted higher values, on the average, than were observed) downstream from the City of Twin Falls STP and Auger Falls (Station S13) and in the area of dense macrophyte beds (Stations S21 and S31). The bias in the total ammonia nitrogen at the station downstream from Auger Falls and from the City of Twin Falls STP could be due to error in the loading from the City of Twin Falls STP. It could also be due to incorrectly characterizing the sources and sinks for total ammonia in the Auger Falls segment of the Study Reach.

Structural errors in the model also may be an important source of bias and poor correlation between simulated and observed total ammonia nitrogen in the vicinity of the macrophyte beds (Stations S21 and S31). This segment of the Study Reach is one of high biological productivity. Uptake and release of ammonia by macrophytes is an important part of the nutrient dynamics in the model. This is evident from the high-frequency components in the simulated values of total ammonia nitrogen (Figures D-12c and D-13c). Additional field and/or laboratory studies would be required to adequately test the hypotheses that control ammonia uptake and release in the model.

D.4.6.4. *Parameter Estimation*

As discussed in Chapter 4, formal solutions of the parameter estimation problem are difficult to obtain. The difficulty in this ecological risk assessment arises from the nonlinear nature of the mass and energy balance equations, limited data, and a large number of parameters. Because of this, parameter estimation was done by trial and error for selected parameters only. The trial-and-error process included initial selection from the literature, followed by adjustment of parameters until simulated values agreed reasonably well with observations. Those parameters estimated by trial and error are listed in Tables D-18 to D-20.

Those parameters included in the trial-and-error process were ones that determine the sedimentation rates of phosphorus; loss, uptake, and release of ammonia nitrogen; growth and death rates of macrophytes and epiphytes; and habitat factors for macrophytes. Model response was particularly sensitive to the parameters for growth and mortality rates of macrophytes and epiphytes and habitat factors for macrophytes. Of particular importance for parameters characterizing growth rates and mortality rates of macrophytes and epiphytes was the role of sediments and sedimentation processes in the Study Reach.

The conceptual model for the macrophytes includes the assumption that nutrient flow from the sediments to the roots of macrophytes is unlimited by plant uptake. This assumption plays a significant role in the development of management strategies for reducing macrophytes in the Study Reach. Given an essentially infinite supply of nutrients, reduction in the discharge of dissolved nutrients is unlikely to result in a decrease in aggregate macrophyte biomass. Although results of macrophyte studies in the Study Reach during 1992 and 1993 (Falter and Carlson, 1994; Falter et al., 1995) suggest that the assumption may not be met in the densest plant beds in late summer, it is a reasonable first approximation for extended periods of low flow when sediment deposition rates are high. However, under conditions of high flow, when sediments are being removed by scouring, this assumption is likely to result in unreasonably high rates of nutrient flow to rooted macrophytes.

The model was also quite sensitive to the magnitude of the habitat factors (Table D-16). The habitat factors, in principle, should be derived from knowledge of sedimentation processes. However, in this case lack of knowledge of these processes made it necessary to use the limited data that were available (Hill, 1992) regarding macrophyte habitat. Although these data provided information about conditions existing during 1990 for certain segments of the Study Reach, they were not sufficient for predicting habitat conditions under various river flow regimes. Recent data (McLaren, 1998) show that habitat factors for macrophytes change dramatically as a result of scouring of sediment deposits and associated macrophytes during periods of high flows in the Snake River. On the basis of his study of sediment transport during July and October 1997, McLaren (1998) concluded these changes occur when the flow in the Study Reach exceeds approximately 10,000 cfs. Such conditions have occurred in the past, but are not reproduced in the cumulative distribution function generated by the model (Figure D-65). Incorporating a predictive model for macrophyte habitat would require more study of the sedimentation processes in the Study Reach.

The simulation results for macrophytes were also sensitive to the parameters used to characterize the physical stress placed on plants by high water velocities. This component of the model had much greater impact on macrophyte density than did changes in the concentration of total phosphorus in the water column. The initial estimates of the parameters used to characterize physical stress from high water velocity were estimated from the work of Chambers et al. (1991a,b). The magnitude of parameters relating macrophyte growth to water velocity was modified to account for the results obtained by McLaren (1998). However, this change was not able to account for the dramatic changes in macrophyte growth observed in 1997. It is likely this is due to the fact that these parameter changes affected only the plant physiology and were not related to the physical processes associated with sediment transport. The physical removal of both the nutrient-laden substrate and the macrophytes through transport processes was the primary factor leading to reduction of macrophyte biomass in the Study Reach (Clark, 1997). The methods described here did not include an analysis of the physical processes characterizing sediment deposition and scouring. Additional studies are necessary to describe these processes and their effects on macrophyte and epiphyte mortality.

D.4.6.5. *State Variable Measurement Variability and Uncertainty*

Field measurements of state variables are needed for testing model hypotheses and, ultimately, evaluating the reliability of the simulation methods. However, there are several sources of variation and uncertainty in field measurements. There is generally some error associated with the instrument or laboratory technique and with the manner in which samples are taken and handled. In general, the largest variability is usually due to spatial or temporal variability of the state variable being measured. This is particularly true for the biological state variables. For example, Falter et al. (1995) and Falter and Burris (1996) observed high spatial variability of macrophyte density at Snake River Mile 600.0 with a high number of replicates. DO is a state variable that can also have high temporal variability during periods of high primary productivity. There will be uncertainty in characterizing the average state of the system's DO when only one grab sample is taken every 2 weeks.

There is also uncertainty in comparing field observations with simulated results because the field observations are point measurements whereas the simulated values are space and time-averaged. The magnitude of the uncertainty depends, of course, on each state variable's spatial and temporal characteristics.

D.4.6.6. *Measurement Endpoints*

Uncertainty and variability in the measurement endpoints can also make significant contributions to variance in the estimates of ecological risk. For this risk assessment, the measurement endpoints are based on the water quality criteria established by the State of Idaho and on habitat suitability indices developed by the USFWS. The quantitative water quality criteria have been developed from a wide range of field and laboratory tests and assays. Such tests often show considerable variability in results. Furthermore, they may not always be appropriate for local conditions because of variability in environmental factors and the response of the target organisms.

In many cases, the exposure period used in the tests or assays used to develop criteria may not be commensurate with exposure periods experienced in the real system. For example, many of the water quality criteria are based on results from bioassays conducted over a period of 48 hours. The results of the bioassays are interpreted in terms of acute or chronic impacts on test organisms. The acute and chronic values are converted to instantaneous or 4-day averages, respectively, in the water body. Interpreting the results of tests and assays in this way is meant to be protective of the ecosystem, but it does introduce uncertainty into the assessment of ecological risk.

There is even more uncertainty associated with the quantitative interpretation of narrative standards. In the Study Reach, the major impact of nutrient and sediment loads is abundant macrophyte growth. There is general agreement that the levels observed during the period 1990-1994 were unacceptable in terms of the narrative standards set by the State of Idaho. However, there is as yet no agreement at the local level on what quantitative measures of macrophyte growth constitute a violation of these standards. Table D-24 represents an effort to develop a measurement endpoint for characterizing nuisance levels of macrophyte biomass. However, the analysis of macrophyte literature is limited in scope and may not necessarily reflect local conditions in the Study Reach. The nuisance criterion should undoubtedly be adjusted for the type of plant community. For example, densities of *P. pectinatus* at a level of 200 g/m^2 might not hinder beneficial uses. Densities of *Cladophora* of this magnitude, because of their surface-floating habit, could severely hinder beneficial uses and therefore be considered a nuisance. Until further discussion and analysis are devoted to this issue, the measurement endpoint for macrophyte biomass should be considered provisional.

The habitat suitability indices, based on the IFIM procedure used by the USFWS (Anglin et al., 1992) in the Swan Falls reach of the Snake River, are valuable as measurement endpoints because they were derived for target organisms found in the Study Reach. They provide measures of ecological impacts on target organisms that are due primarily to physical stressors. These measures have been developed for various life stages of the target organisms and can be used to evaluate the time history of impacts to these life stages. There is uncertainty in the values of these indices, however. The uncertainty is due mostly to lack of specific knowledge regarding impacts of the stressors on the target organisms. Some of these habitat suitability indices, for example, are based on surveys of the best professional judgment of regional fisheries biologists. The results of the surveys were converted to indices as part of the IFIM (Anglin et al., 1992) without quantifying the variability and uncertainty in the responses.

Because the focus of IFIM is on instream flows, the habitat suitability indices give considerable weight to the physical parameters, depth and velocity. Some researchers (e.g., Mathur et al., 1985) have reported there is little evidence showing correlation between the habitat suitability indices and fish biomass. Proponents of the methodology (e.g., Armour and Taylor, 1991) acknowledge the lack of post-project monitoring to document the value of IFIM, while noting that IFIM has become widely accepted as a tool for investigating the impacts of flow regulation on aquatic resources. The rationale for adapting the IFIM methodology for ecological risk assessment in the Study Reach was based primarily on this acceptance. However, given the manner in which the indices were developed and the lack of documented support for the method, the results reported here are likely to have a high degree of uncertainty.

APPENDIX D-1
Equations of Mass and Energy Conservation for Surface Waters

APPENDIX D-1. EQUATIONS OF MASS AND ENERGY CONSERVATION FOR SURFACE WATERS

The general equation (Equation D2-1) contains terms for advection, turbulent diffusion, and source/sink terms for physical, chemical, and biological processes. The advection and turbulent diffusion processes are the same for all constituents and are described in Section 2.1.1. This appendix contains the conceptual models and the equations for the source/sink terms in the equations of conservation of mass and energy in surface waters.

D.1.1. Carbonaceous Biological Oxygen Demand (CBOD), C_1

When microorganisms metabolize organic matter in rivers, lakes, and streams, they use dissolved oxygen as they reduce the organic carbon to carbon dioxide. The available demand for dissolved oxygen associated with organic carbon is characterized as carbonaceous oxygen demand (CBOD). The rate at which dissolved oxygen is consumed is assumed to be a linear function of the mass of available CBOD and that one unit of CBOD is metabolized for each unit of dissolved oxygen.

The source/sink terms in the conservation equations for CBOD can be written as:

$$\Gamma_{1j} = K_1 C_1 V_j \qquad \text{(D1-1)}$$

where

$$K_1 = K_1^{20} \; \theta_{1T}$$

$K_1^{20} =$ the deoxygenation rate at 20 °C, days^{-1}

$\theta 1T$ = the temperature correction factor for deoxygenation.

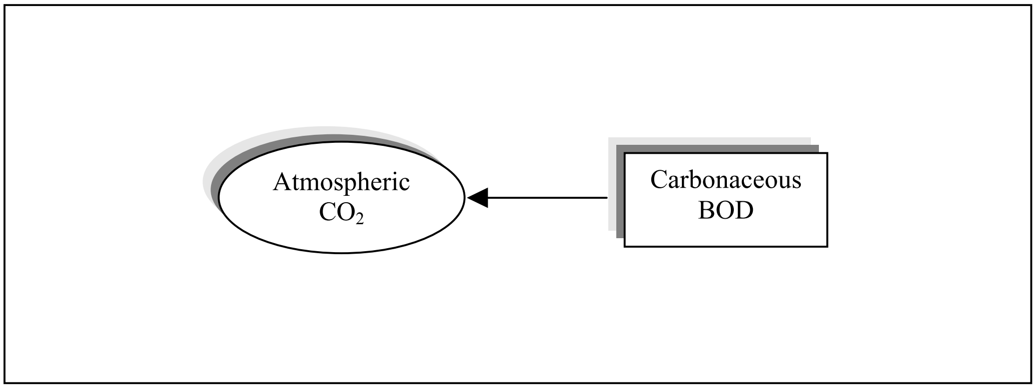

Figure A-1. Conceptual model for sources and sinks of carbonaceous biological demand (CBOD).

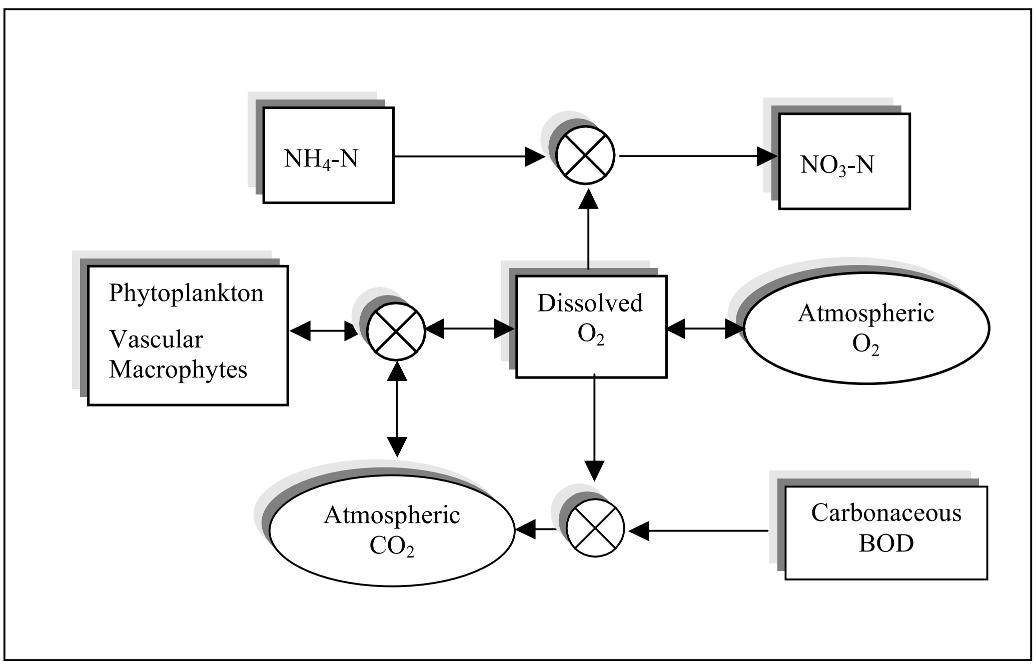

Figure A-2. Conceptual model for sources and sinks of dissolved oxygen (DO).

D.1.2. Dissolved Oxygen (DO), C_2

The major source and sink terms for dissolved oxygen (DO) include the following:

- Oxygen used by microorganisms to metabolize organic matter (CBOD),
- Oxygen used by nitrifying microorganisms to convert ammonium to nitrate (nitrogenous biological oxygen demand, NBOD),
- Respiration of phytoplankton, and
- Photosynthesis by phytoplankton.

The mass balance is

$$\Phi_{2j} = \left[K_2 (C_{sat} - C_2) + \alpha_{oc} G_{\pi} C_3\right] V_j + B_{photo} \qquad \text{(D1-2)}$$

$$\Gamma_{2j} = (K_1 C_1 + \alpha_{on} K_{56} C_3) V_j + B_{resp} \qquad \text{(D1-3)}$$

where

K_2 = the reaeration rate, days^{-1},

APPENDIX D-2
Formulation of the Mathematical Model for Benthic Plants

APPENDIX D-2. FORMULATION OF THE MATHEMATICAL MODEL FOR BENTHIC PLANTS

The equations describing the rates of changes of benthic plant biomass due to internal sources and sinks are developed below.

D.2.1. Macrophyte Root Biomass

The growth of root biomass is controlled by a basic growth rate, which is a function of the ambient temperature, the quantity of labile carbon stores, and the level of habitat affected by crowding. Endogenous respiration is controlled by temperature only. The model assumes no grazing by herbivores or mortality due to scouring.

$$\frac{dM_R}{dt} = \mu_R\, M_R - \rho_R\, M_R \qquad \text{(D2-1)}$$

where

M_R = macrophyte root biomass, gm/m^2;

$$\mu_R = \mu_{R_{max}}\ \mathrm{MAX}\left(\frac{M_C - M_{C_0}}{K_{M_C} + M_C - M_{C_0}}, 0\right)\left(\frac{M_{R_{max}}}{M_{R_{max}} + \sum M_R}\right) f_R^{\mu}(T),\ \text{days}^{-1};$$

$$f_R^{\mu}(T) = e^{-2.3\left(\frac{T - T_{opt}}{T_{max} - T_{opt}}\right)^2} \qquad T \geq T_{opt}$$

$$= e^{-2.3\left(\frac{T - T_{opt}}{T_{min} - T_{opt}}\right)^2} \qquad T \leq T_{opt}$$

$$\rho_R = \rho_{R\,max}\ \mathrm{MAX}\left(\frac{M_{C_0} - M_C}{|M_{C_0} - M_C|}, 0\right) f_R^{\rho}(T),\ \text{days}^{-1},$$

$$f_R^{\rho}(T) = Q_{R10}^{\rho\ (T-20)}$$

M_{C_0} = minimum carbohydrate stores for root or shoot/leaf growth, gm/m^2,

$M_{R_{max}}$ = maximum concentration of macrophyte root mass, $gm/meter^2$,

K_{M_C} = level of carbohydrate store for which growth levels are ½ maximum, gm/m^2.

D.2.2. Macrophyte Shoot/Leaf Biomass

The growth of shoot/leaf biomass is also controlled by a basic growth rate, which is a function of the ambient temperature, the quantity of labile carbon stores, and the level of habitat affected by crowding. Endogenous respiration is controlled by temperature only. The model accounts for mortality due to high river velocities, but assumes there is no grazing by herbivores.

$$\frac{dM_L}{dt} = \mu_L\, M_L - \rho_L\, M_L - D_L - G_L \qquad \text{(D2-2)}$$

M_L = macrophyte leaf/shoot biomass, gm/m^2,

$$\mu_L = \mu_{L_{max}}\ \mathrm{MAX}\left(\frac{M_C - M_{C_0}}{K_{M_C} + M_C - M_{C_0}}, 0\right)\left(\frac{M_{L_{max}}}{M_{L_{max}} + \sum M_L}\right) f_L^{\mu}(T)\ f_L^{\mu}(L)\ ,\text{days}^{-1}$$

$f_L^{\mu}(L)$= Biomass limitation for leaf/shoot growth, dimensionless,

$$f_L^{\mu}(T) = e^{-2.3\left(\frac{T - T_{opt}}{T_{max} - T_{opt}}\right)^2} \qquad T \geq T_{opt}$$

$$= e^{-2.3\left(\frac{T - T_{opt}}{T_{min} - T_{opt}}\right)^2} \qquad T \leq T_{opt}$$

$$\rho_L = \rho_{L_{max}} \ \mathrm{MAX}\left(\frac{M_{C_0} - M_C}{| M_{C_0} - M_C |}, 0\right) f_L^{\rho}(T) \ \text{, days}^{-1},$$

$$f_L^{\rho}(T) = Q_{L10}^{\rho\ (T-20)},$$

$$D_L = \delta_{L_{max}} \left(f_L^{\delta}\ (\text{Season}) f_L^{\delta}(T)\right) M_L \text{, gm/m}^2\text{ / day},$$

$M_{R_{max}}$ = maximum concentration of macrophyte root mass, gm/m^2,

$\delta_{L_{max}}$ = maximum mortality rate for leaf/shoots, days^{-1},

f_L^{δ}(Season) = seasonal function for leaf/shoot mortality, dimensionless,

= 0 if Julian Data >240 and water temperature, T < 15.0 C,

= 1 otherwise.

$f_L^{\delta}(T)$ = 1 - $f_L^{\rho}(T)$, dimensionless,

ρ_L^V = maximum mortality rate for leaf/shoots from river velocity, days^{-1},

D.2.3. Macrophyte Carbohydrate Pool

In this model, as net organic carbon is produced during photosynthesis it is stored as labile carbon. Carbon from this pool is used for roots and shoot/leaf biomass. Carbon stores are also lost during respiration.

$$\frac{dM_C}{dt} = \pi_C\, M_L - \rho_L\, M_L - \rho_R\, M_R - \left(\frac{\mu_L\, M_L + \mu_R\, M_R}{\varepsilon_C}\right) \qquad \text{(D2-3)}$$

M_C = carbohydrate pool, gm/m^2,

$$\pi_C = \pi_{C\max}\, f_I^{\pi}(I_0, z, \kappa)\, f_T^{\pi}(T)\, f_N^{\pi}(N)\, f_V^{\pi}(V)$$

$\pi_{C\max}$ = maximum photosynthetic rate, days^{-1},

$$f_I^{\pi}(I_0, z, \kappa) = \frac{f}{\kappa(z_2 - z_1)} \ln\left(\frac{K_I^{\pi} + I_0\, e^{-\kappa z_1}}{K_I^{\pi} + I_0\, e^{-\kappa z_2}}\right)$$

I_0 = photosynthetically active solar radiation, kcal/m^2/second,

K_I^{π} = limiting, photosynthetically active solar radiation, kcal/m^2/second,

k = light extinction coefficient, meters^{-1},

z = water depth, meters,

$$f_T^{\pi}(T) = e^{-2.3\left(\frac{T - T_{opt}}{T_{max} - T_{opt}}\right)^2} \qquad T \geq T_{opt}$$

$$= e^{-2.3\left(\frac{T - T_{opt}}{T_{min} - T_{opt}}\right)^2} \qquad T \leq T_{opt}$$

$$f_N^{\pi}(N) = \mathrm{MIN}\left(\frac{\sum N}{K_N^{\pi} + \sum N}, \frac{\sum SRP}{K_{SRP}^{\pi} + \sum SRP}\right), \text{ dimensionless,}$$

6N = the sum of dissolved inorganic forms of nitrogen, mg/L,

6SRP = the sum of dissolved inorganic forms of phosphorus, mg/L,

K_N^{π} = the Michaelis-Menten constant for inorganic nitrogen, mg/L,

K_{SRP}^{π} = the Michaelis-Menten constant for inorganic phosphorus, mg/L,

H_C = efficiency of converting from labile stores to roots and leaf/shoot, dimensionless.

D.2.4. Epiphyte Biomass

The growth rate of epiphytic algae is a function of the available depth-averaged light, the ambient temperature, the concentration of inorganic phosphorus and nitrogen, and the velocity. At low velocities, the growth rate is reduced to account for limited nutrient supply. At high velocities the growth is reduced as the epiphytes are physically stressed. Respiration rates are a function of ambient temperature. The model assumes there is no mortality due to grazing.

$$\frac{dM_E}{dt} = \mu_E M_E - \rho_E M_E - D_E \qquad \text{(D2-4)}$$

M_E = the biomass of epiphytes, gm/m^2,

$$\mu_E = \mu_{E\,max} \; f_I^{\mu}(I_0, z, \kappa)\; f_T^{\mu}(T)\; f_N^{\mu}(N)\, f_V^{\mu}(V) f_L^{\mu}(L)$$

μ_{Emax} = the maximum growth for epiphytes, days^{-1},

$$f_I^{\pi}(I_0, z, \kappa) = \frac{f}{\kappa(z_2 - z_1)} \ln\left(\frac{K_I^{\pi} + I_0\, e^{-\kappa z_1}}{K_I^{\pi} + I_0\, e^{-\kappa z_2}} \right)$$

$$\mu_P = \mu_{Pmax}\, f_I^{\pi}(I_0, z.\kappa)\, f_T^{\mu}(T)\, f_N^{\mu}(N)\, f_V^{\mu}(V)$$

$$f_T^{\mu}(T) = e^{-2.3\left(\frac{T - T_{opt}}{T_{max} - T_{opt}}\right)^2} \qquad T \geq T_{opt} \quad \text{, dimensionless,}$$

$$= e^{-2.3\left(\frac{T - T_{opt}}{T_{min} - T_{opt}}\right)^2} \qquad T \leq T_{opt}$$

$$f_N^{\mu}(N) = \mathrm{MIN}\left(\frac{\sum N}{K_N^{\mu} + \sum N}, \frac{\sum SRP}{K_{SRP}^{\mu} + \sum SRP} \right) \text{, dimensionless,}$$

$$f_V^{\mu}(V) = 4\left(\frac{V}{V_{max}^{\mu}}\right)\left(1 - \frac{V}{V_{max}^{\mu}}\right) \text{, dimensionless,}$$

$$f_L^{\mu}(L) = \frac{\sum L}{\sum L + K_L^{\mu}}$$

$\sum L$ = the total biomass of macrophyte leaves, gm/m^2,

K_L^{μ} = leaf biomass for which optimal epiphyte growth rate is 1/2 the maximum, gm/m^2,

$$\rho_E = \rho_{E\max} \; f_E^{\rho}(T)$$, days^{-1},

$\rho_{E\max}$ = the maximum ephiphyte respiration rate, days^{-1},

$$f_E^{\rho}(T) = Q_{E10}^{\rho\ (T-20)}$$, dimensionless.

D.2.5. Periphyton Biomass

The growth rate of periphyton is a function of the available depth-averaged light, the ambient temperature, the concentration of inorganic phosphorus and nitrogen, and the velocity. At low velocities the growth rate is reduced to account for limited nutrient supply. At high velocities the growth is reduced as the periphyton are physically stressed. Respiration rates are a function of ambient temperature. The model assumes there is no mortality due to grazing.

$$\frac{dM_P}{dt} = \mu_P M_P - \rho_P M_P - D_P \qquad \text{(D2-5)}$$

M_P = the biomass of periphyton, gm/m^2,

$$\mu_P = \mu_{P\max} \; f_I^{\pi}(I_0, z.\kappa) f_T^{\mu}(T) f_N^{\mu}(N) f_V^{\mu}(V)$$

$\mu_{P\max}$ = the maximum growth for periphyton, days^{-1},

$$f_I^{\pi}(I_0, z, \kappa) = \frac{f}{\kappa(z_2 - z_1)} \ln\left(\frac{K_I^{\pi} + I_0\, e^{-\kappa z_1}}{K_I^{\pi} + I_0\, e^{-\kappa z_2}}\right)$$

, dimensionless,

$$f_T^{\mu}(T) = e^{-2.3\left(\frac{T - T_{opt}}{T_{max} - T_{opt}}\right)^2} \qquad T \geq T_{opt}$$

$$= e^{-2.3\left(\frac{T - T_{opt}}{T_{min} - T_{opt}}\right)^2} \qquad T \leq T_{opt}$$, dimensionless,

$$f_N^{\mu}(N) = MIN\left(\frac{\sum N}{K_N^{\mu} + \sum N}, \frac{\sum SRP}{K_{SRP}^{\mu} + \sum SRP}\right) \quad f_V^{\mu}(V) = 4\left(\frac{V}{V_{max}^{\mu}}\right)\left(1 - \frac{V}{V_{max}^{\mu}}\right)$$, dimensionless,

$$\rho_P = \rho_{P_{max}} \; f_P^{\rho}(T)$$, days $^{-1}$,

$\rho_{P_{max}}$ = the maximum periphyton respiration rate, days^{-1},

$$f_P^{\rho}(T) = Q_{P10}^{\rho\;(T-20)}$$, dimensionless

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